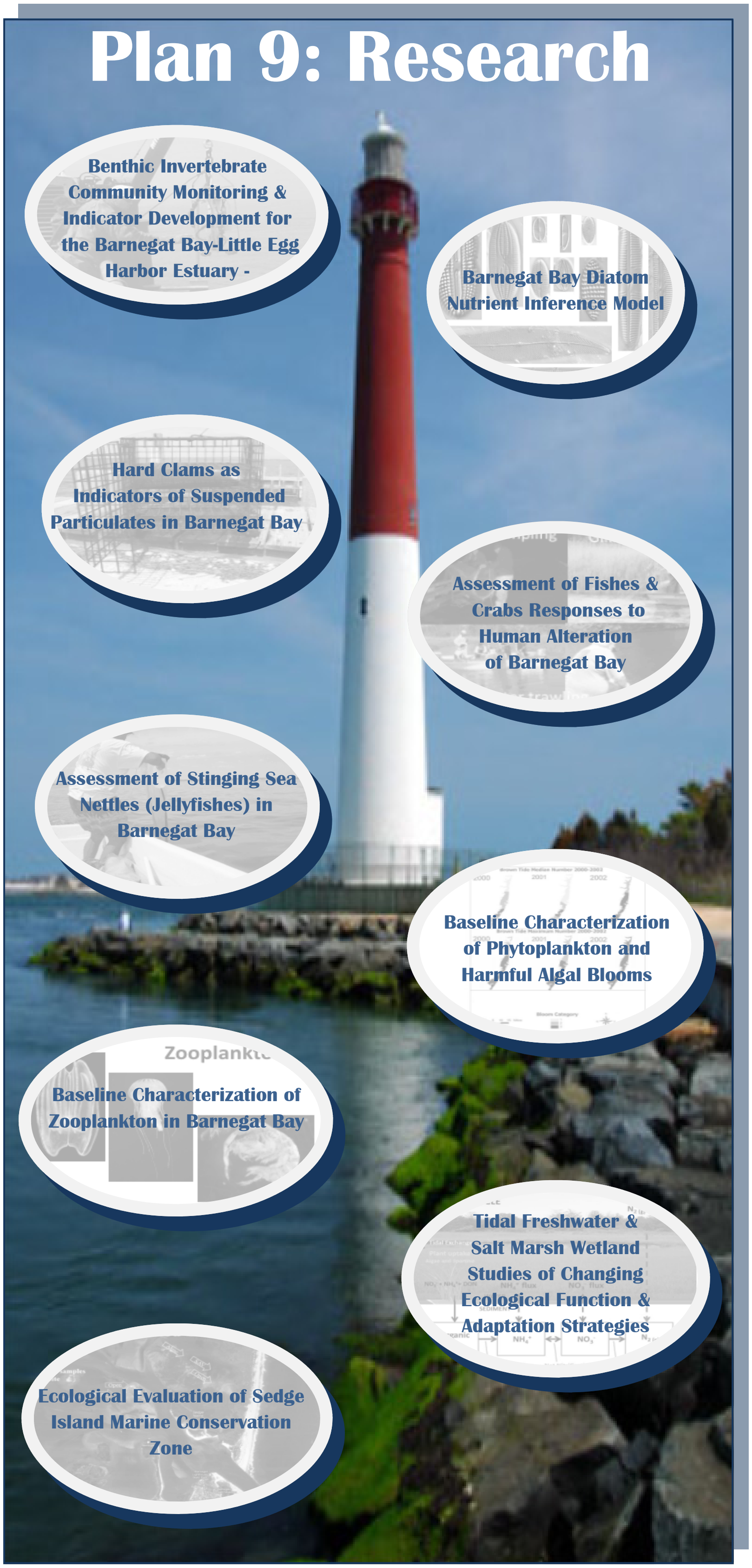
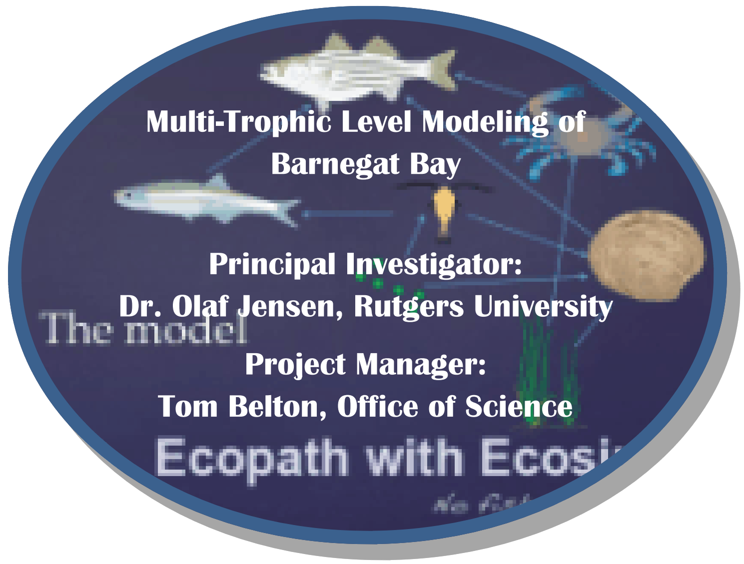


Barnegat Bay—Year 2



Thomas Belton, Barnegat Bay Research Coordinator
Dr. Gary Buchanan, Director—Office of Science
Bob Martin, Commissioner, NJDEP
Chris Christie, Governor

June, 2015

Year 2 Project Report for “Multi-Trophic Level Modeling of Barnegat Bay”

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Objective 1: Refine conceptual model developed through interviews with Barnegat Bay scientists and managers

The conceptual model developed through stakeholder interviews during Year 1 of the project was further refined and analyzed during the first half of the current grant period and a manuscript was submitted to the Department for their review. Approval for submission was obtained from the Department on March 11, 2014, and the manuscript was subsequently sent to the Journal of Environmental Management. The manuscript received brief but supportive comments from a single reviewer. We revised the manuscript accordingly and it is currently under review at the journal. The submitted version is included here as Appendix 4.

Objective 2 - Incorporate Year 1 data and model parameters from NJDEP funded projects for use in the NPZ and EwE models

EwE model parameters

Ecopath with Ecosim (EwE) is a software modeling tool used to quantitatively evaluate trophic interactions within an ecosystem in order to assess options for ecosystem-based management of fisheries. The first step in the process is to develop a mass-balance model (Ecopath), which requires four groups of basic input parameters to be entered into the model for each of the species (or groups) of interest: diet composition, biomass accumulation, net migration, and catch (for fished species). Three of the following four additional input parameters must also be input: biomass, production/biomass (Z), consumption/biomass, and ecotrophic efficiency. The model uses the input data along with algorithms and a routine for matrix inversion to estimate any missing basic parameters so that mass balance is achieved.

For the purposes of the Barnegat Bay model we have set biomass accumulation and net migration to zero for all of our species groups. This is equivalent to the assumption that biomass of all species groups was at equilibrium. This is a typical assumption in the absence of information to the contrary. The biomass, production/biomass, consumption/biomass, and Ecotrophic efficiency values for the model can be found in Table 1 below. These parameters were estimated from a variety of sources, the details of which can be found in Appendix 1. The diet composition matrix can be found in Appendix 2, with the source data also listed in Appendix 1. Harvest data for recreationally and commercially important species can also be incorporated into the EcoPath model as the landings (t/km^2/year) for the year in which the model is initiated. The landings values included in the model can be found in Table 2 below, with commentary on their derivations found in Appendix 3.

As identified in Appendix 1, many of the parameters utilized in the model at this time were not developed specifically for Barnegat Bay. This is particularly true for the biomass estimates, where with the exceptions of SAV, hard clams, bay anchovy, sea nettles, and ctenophores, the other values were primarily estimated by the software, or modified from Chesapeake Bay values (seabirds). The SAV, hard clam, and bay anchovy biomass values were from studies conducted around the time of the initial year of the model. The ctenophore and sea nettle biomasses were estimated using data from the first year of the NJDEP Barnegat Bay field research projects. We attempted to utilize data from the first year of the NJDEP Barnegat Bay field research projects in combination with other Barnegat Bay specific studies for phytoplankton and amphipods but the biomass estimated by these studies was substantially less than that required to support the remainder of the model. We will revisit these estimates as additional years of Barnegat Bay specific data become available. We are also in the process of completing a simple stock assessment model to estimate biomass for blue crab given their importance to the recreational and commercial fishery sector. If successful this value will be utilized in place of the software derived estimate.

Table 1: Basic parameters for the Barnegat bay Ecosystem Model. Values estimated by Ecopath are shown in *italics*. Estimated from a variety of sources as described in Appendix 1.

Group name	Biomass (t/km^2)	Prod./biomass ($year^{-1}$)	Cons./biomass ($year^{-1}$)	Ecotrophic Efficiency	Prod./Cons.
Piscivorous seabirds	0.250	0.163	120	*0.0*	*0.001*
Non-piscivorous seabirds	0.121	0.511	120	*0.0*	*0.004*
Weakfish	*4.472*	0.260	3	0.9	*0.087*
Striped bass	*1.642*	0.4	2.4	0.9	*0.167*
Summer flounder	*2.3*	0.52	2.6	0.95	*0.200*
Bluefish	*2.733*	0.52	3.1	0.95	*0.168*
Winter flounder	*4.661*	0.52	3.4	0.95	*0.153*
Atlantic silversides	*4.741*	0.8	*4*	0.95	0.2
Atlantic croaker	*0.196*	0.916	4.2	0.9	*0.218*
Spot	*0.617*	0.9	6.2	0.9	*0.145*
Atlantic menhaden	*12.697*	0.5	31.42	0.95	*0.016*
River herring	*1.180*	0.75	8.4	0.95	*0.089*
Mummichog	*3.465*	1.2	3.65	0.95	*0.329*
Bay anchovy	4.860	3	9.7	*0.98*	*0.309*
Benthic infauna/epifauna	*81.025*	2	*10*	0.9	0.2
Amphipods	*3.438*	3.8	*19*	0.9	0.2
Blue crabs	*6.366*	1.21	4	0.95	*0.303*
Hard clams	26.18	1.681	5.1	*0.185*	*0.330*
Oyster	0.001	0.630	2	*0*	*0.315*
Copepods	*15.505*	25	*83.333*	0.95	0.3
Microzooplankton	*8.343*	140	*350*	0.95	0.4
Sea nettles	1.380	13	20	*0.077*	*0.650*
Ctenophores	7.860	16.2	35	*0.114*	*0.463*
Benthic algae	*4.614*	80		*0.900*	
Phytoplantkon	*25.221*	160		0.95	
SAV	5.820	5.11		*0.105*	
Detritus	1			*0.110*	

Table 2: Landings values used in the 1981 Barnegat Bay Ecopath model. All values are in tons/km^2/yr. Sources and calculations can be found in Appendix 3.								
Group name	crab - recreational	crab pot and trap	crab winter dredge	commercial clam	OCNGS	jellyfishers	weakfish	striped bass
Piscivorous seabirds								
Non-piscivorous seabirds								
Weakfish					0.026182		0.01208	
Striped bass								0.0931
Summer flounder					0.001699			
Bluefish					2.15E-05			
Winter flounder					0.007052			
Atlantic silversides					0.024835			
Atlantic Croaker					0.013108			
Spot								
Atlantic Menhaden					0.057949			
River herring					0.000742			
Mummichog					7.00E-07			
Bay anchovy					0.011175			
Benthic infauna/epifauna								
Amphipods								
Blue crabs	0.634767	0.656989	0.136559		0.011571			
Hard clams				0.8129				
Oyster								
Copepods								
Microzooplankton								
Sea nettles						1.38		
Ctenophores								
Benthic algae								
Phytoplantkon								
SAV								
Detritus								
Sum	0.634767	0.656989	0.136559	0.8129	0.154333	1.38	0.01208	0.0931

Table 2 cont'd: Landings values used in the 1981 Barnegat Bay Ecopath model. All values are in tons/km^2/yr. Sources and calculations can be found in Appendix 3.								
Group name	summer flounder	bluefish	winter flounder	croaker	spot	menhaden	river herring	Total
Piscivorous seabirds								
Non-piscivorous seabirds								
Weakfish								0.038262
Striped bass								0.0931
Summer flounder	0.804717							0.806416
Bluefish		0.750072						0.750094
Winter flounder			0.9253					0.932352
Atlantic silversides								0.024835
Atlantic Croaker				0.0001				0.013108
Spot					0.00398			0.00398
Atlantic Menhaden						0.000716		0.058665
River herring							0.000358	0.0011
Mummichog								7.00E-07
Bay anchovy								0.011175
Benthic infauna/epifauna								0
Amphipods								0
Blue crabs								1.439886
Hard clams								0.8129
Oyster								0
Copepods								0
Microzooplankton								0
Sea nettles								1.38
Ctenophores								0
Benthic algae								0
Phytoplantkon								0
SAV								0
Detritus								0
Sum	0.804717	0.750072	0.9253	0	0.00398	0.000716	0.000358	6.365871

EwE time series data

Once the Ecopath model has been balanced the mass-balanced linear equations are then re-expressed as coupled differential equations so that they can be used by the Ecosim module to simulate what happens to the species groups over time (Christensen and Walters, 2004). Model runs are compared with time-series data and the closest fit is chosen to represent the system. Time-series data for model calibration are thus essential for developing and validating an Ecosim model (Christensen *et al*. 2009). Therefore, time-series data depicting trends in relative and absolute biomass, fishing effort by gear type, fishing and total mortality rates, and catches for as long a period as possible should be viewed as additional data requirements.

In addition to the commercial and recreational landings information as described in Appendix 2 there are few other time-series data available specific to Barnegat Bay. Many other ecosystem models glean data from formal stock assessments, which utilize similar time series data for single species management plans. Unfortunately there are no stock assessments specific to the Barnegat Bay. We have utilized the commercial blue crab landings data gathered by the NJDEP to create gear specific time series which were converted to effort and used to force the model. We are in the process of completing a simple stock assessment model to estimate a time series of biomasses for blue crab specific to Barnegat Bay, and will include that as a separate time series if successful.

We have acquired a long-term (1988-2011, except 1991-1995) otter trawl data set from the Rutgers Marine Field Station that includes 6 regularly sampled sites located in Little Egg Harbor. The CPUEs generated from this data are useful for fitting to overall trends. We have performed a trawl efficiency study for the DEP sponsored survey, which utilizes the same gear as this survey. Trawl efficiency estimates account for the fact that not all individuals within the path of the trawl are captured. Efficiency estimates will allow us to develop baywide biomass estimates from the current survey data, which we can then use to fit the time series endpoints. The results of the trawl efficiency study are presented here under Objective 3 revised.

In addition to the fish and crab data referenced above, the NJDEP has hard clam surveys from 1986/1987 in Barnegat Bay and Little Egg Harbor, 2001 in Little Egg Harbor, 2011 in Little Egg Harbor, and 2012 in Barnegat Bay. The 1986/1987, 2001, and 2011 data are incorporated into the model. Release of the

2012 data was delayed due to the effects of Hurricane Sandy. This data will be incorporated when it becomes available.

SAV coverage for the bay is available for 1980, 1987, 1999, 2003, and 2009 based on aerial photograph analysis in Lathrop et al. (2001) and Lathrop and Haag (2011). The acreage of seagrass in each year serves as a datapoint of relative abundance. Limited data was available for benthic algae and a time series was not able to be developed.

The last source of Barnegat Bay specific time series data comes from OCNGS. Because of the nature of OCNGS operations, the cooling and dilution intake structures function as an on/off type activity, with the only shutdowns associated with temporary, short term maintenance. As such the plant flow is fairly consistent, and therefore the impacts of the plant can be modeled as a steady forced effort.

An additional source of fish time series data incorporated into the model is an index of biomass generated from the near-shore trawl surveys conducted each fall by the NJDEP. While sampling for this survey occurs along the entire New Jersey coast, it provides an estimate of relative biomass in each year for those species that leave the estuary each fall for offshore or southern waters.

EwE model

The Ecopath model shown in Figure 1 represents a possible configuration of Barnegat Bay for 1981, with the groups arranged by trophic level. There are no surprises in the trophic level of any of the groups, though striped bass in our system do occupy a slightly higher level than those in the Chesapeake Bay. The fact that this model output is parsimonious with other models of similar systems lends additional support to its interpretation. The model is balanced, in that there is sufficient food for the consumers and enough production to meet consumptive demands.

When the time series data is incorporated into the model and the vulnerability values are adjusted to fit to the time series , the overall fit of the model prediction to the available data is reasonable (Figure 2, Sum of Squares = 487.1). The model fits most of the groups well, with changes in relative biomass from the time series data reflected in the model (Figure 3). The increase in relative biomass of croaker throughout the time series is reflective of the increase in its overwintering survivability and general population increase in the Mid-Atlantic as documented by Hare and Able (2007). However the biomass and catch values, particularly the OCNGS catches, appear to be somewhat inflated and warrant further investigation and refinement.

This EcoSim run includes forcing functions for benthic algae and submerged aquatic vegetation (SAV) in an effort to replicate changes in primary producers over time (Figure 4). The benthic algae forcing function is a nearly linear increase from 1981 to 2000 and then no increase for the remainder of the time series, with a 1.5x increase from the beginning to the end of the time series. The SAV function is a steady decrease over the time series to about half of the original. These rates are an estimate of forcing based on the historic decline in SAV and the anecdotal increase in benthic macroalgae.

Figure 1: Barnegat Bay 1981 model. Numbered horizontal lines indicate trophic level.

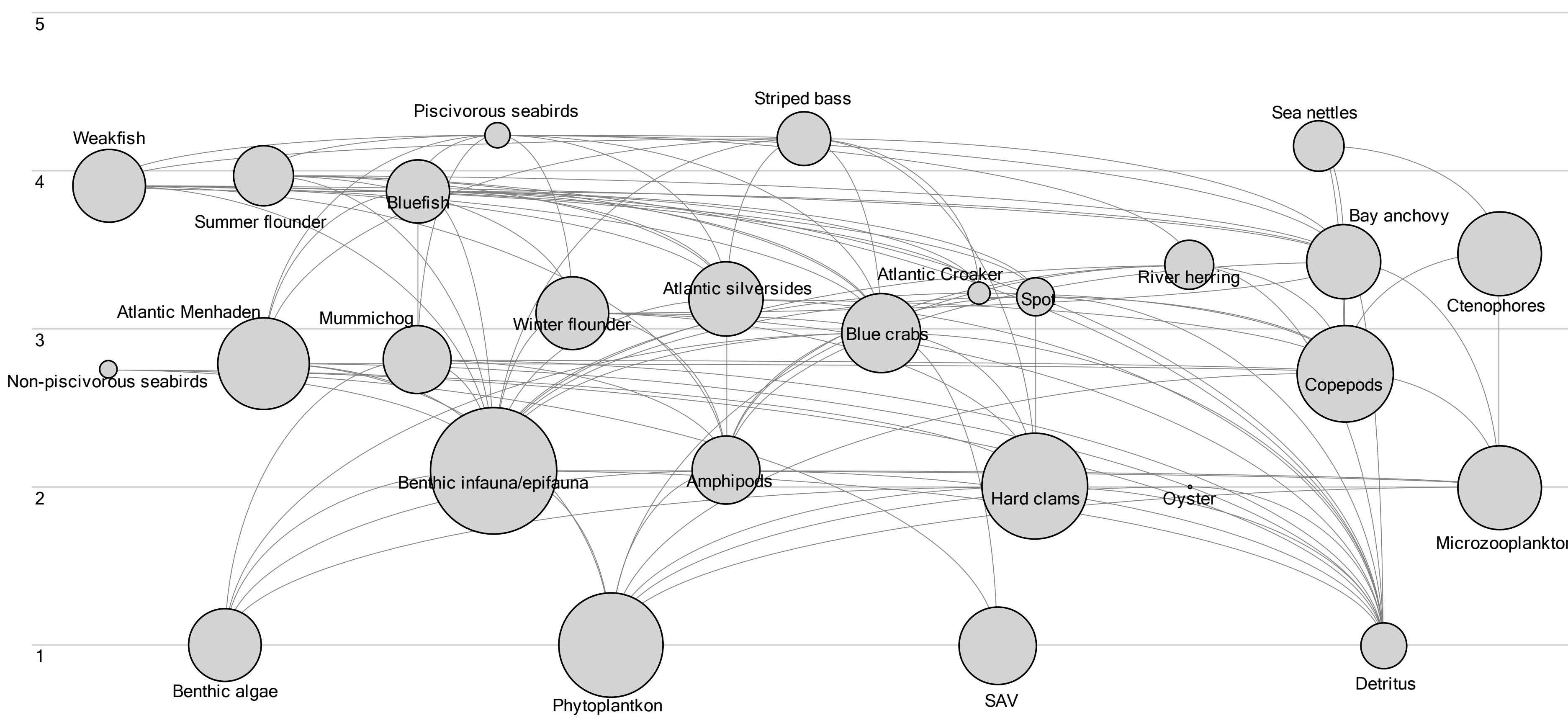
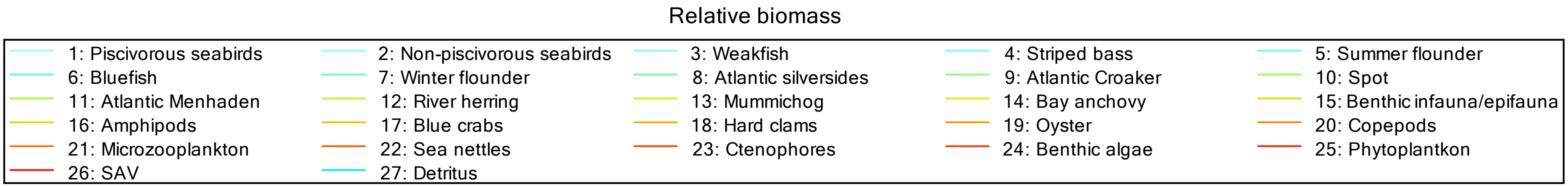
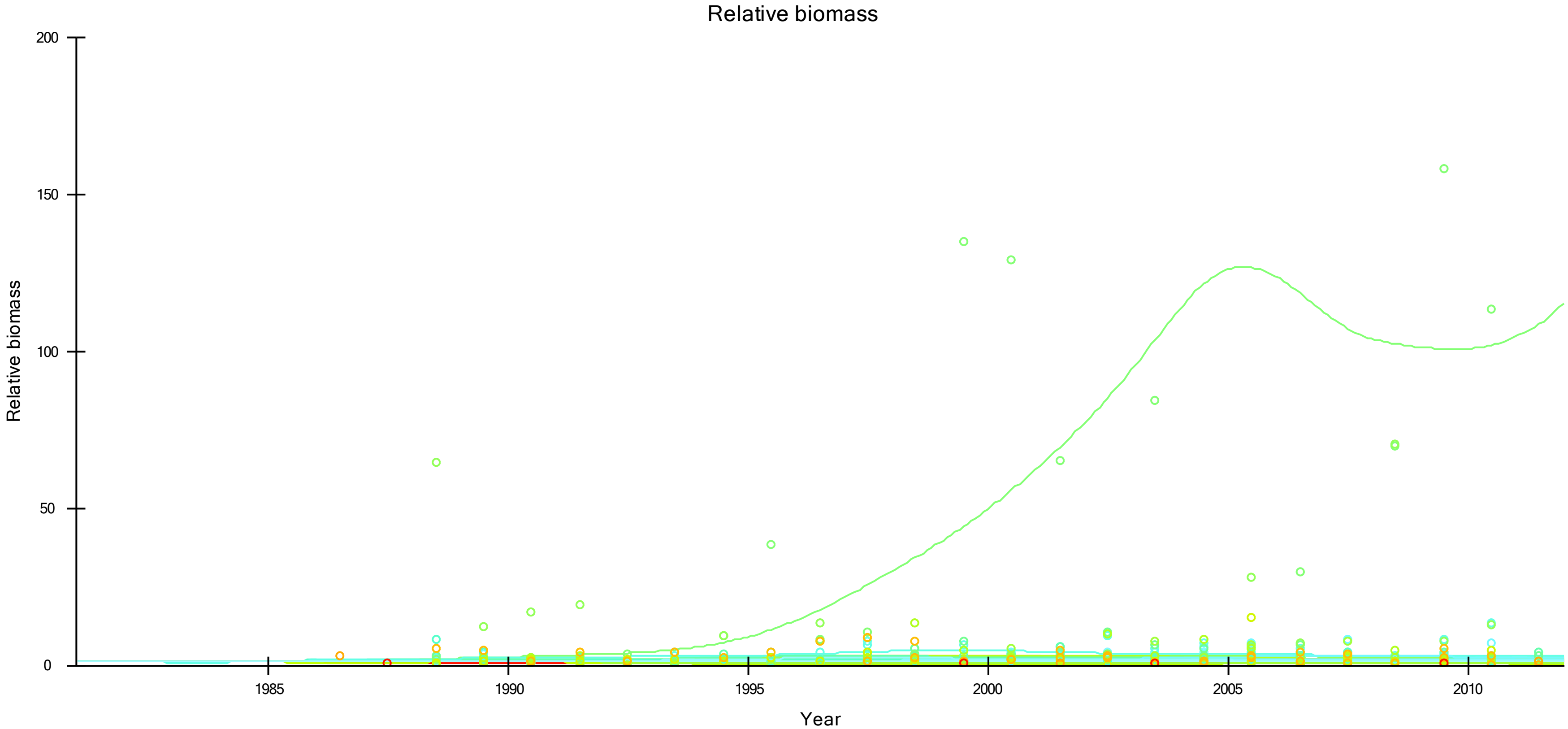


Figure 2: Model predictions versus time series data for 1981 through 2012.





Relative biomass
200
150
100
50
0
Relative biomass
1985
1990
1995
2000
2005
2010
Year

Figure 3: Graphs of the model fit to the currently available time series data for each of the groups in the EwE model.

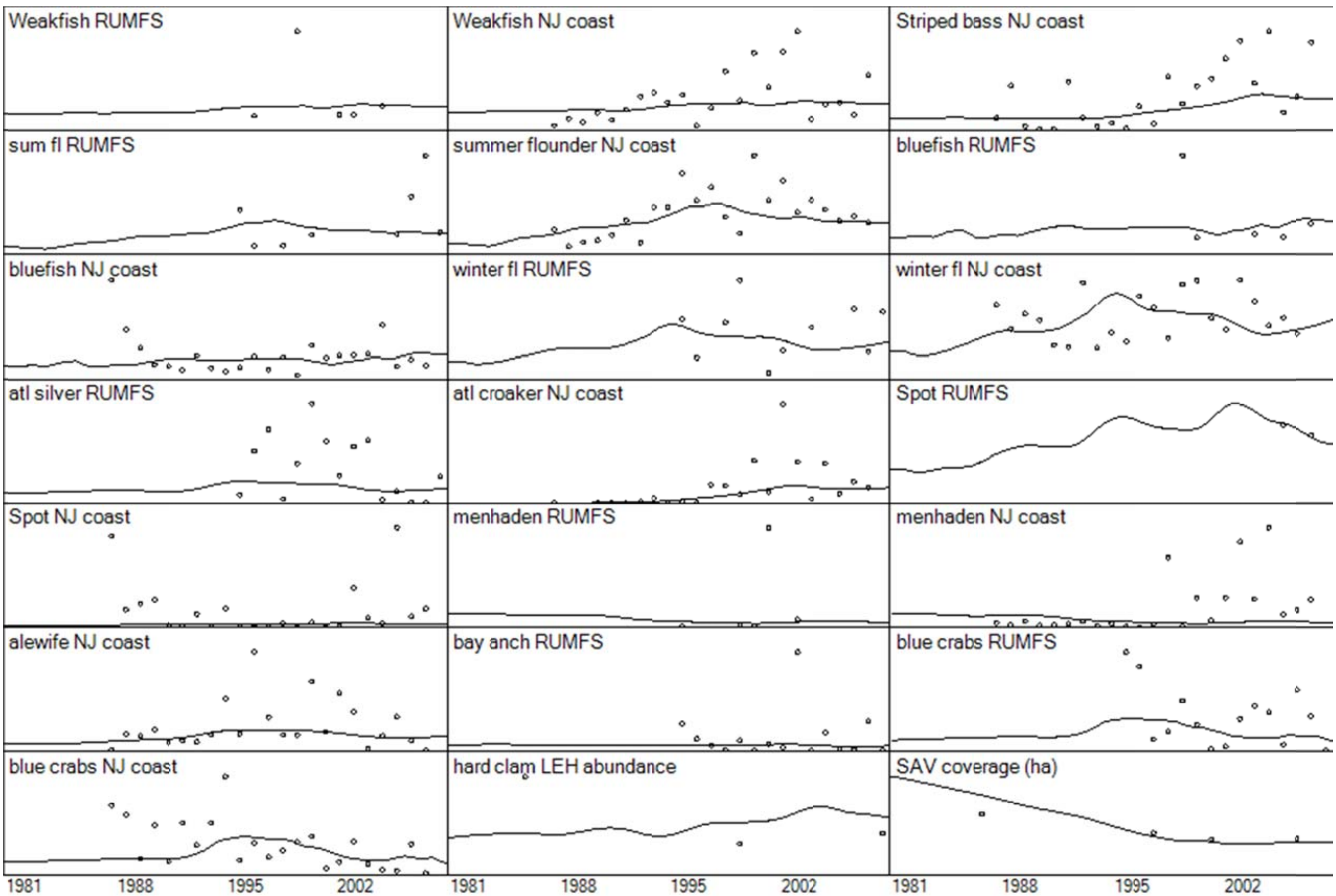
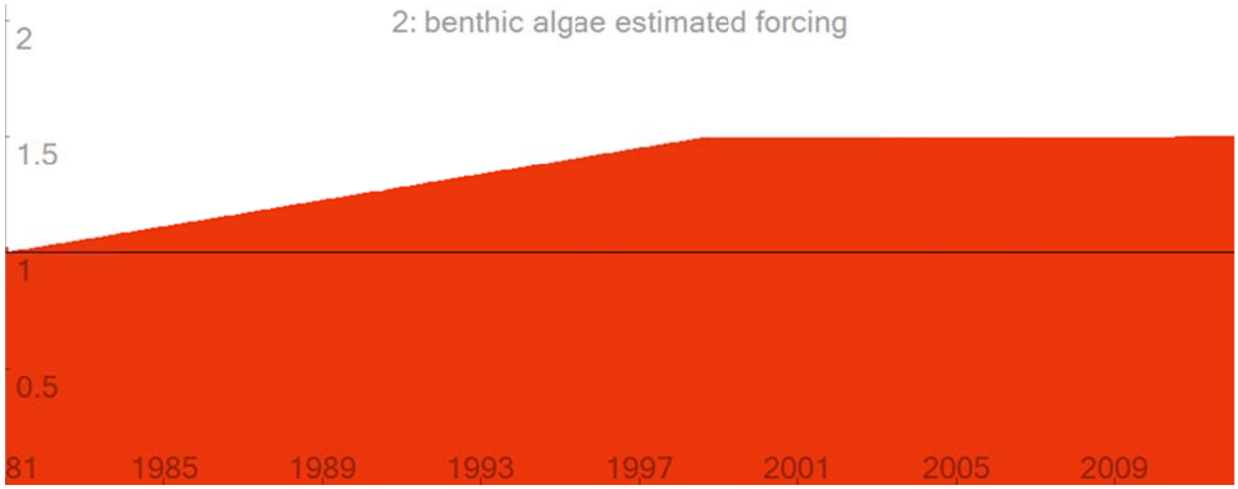
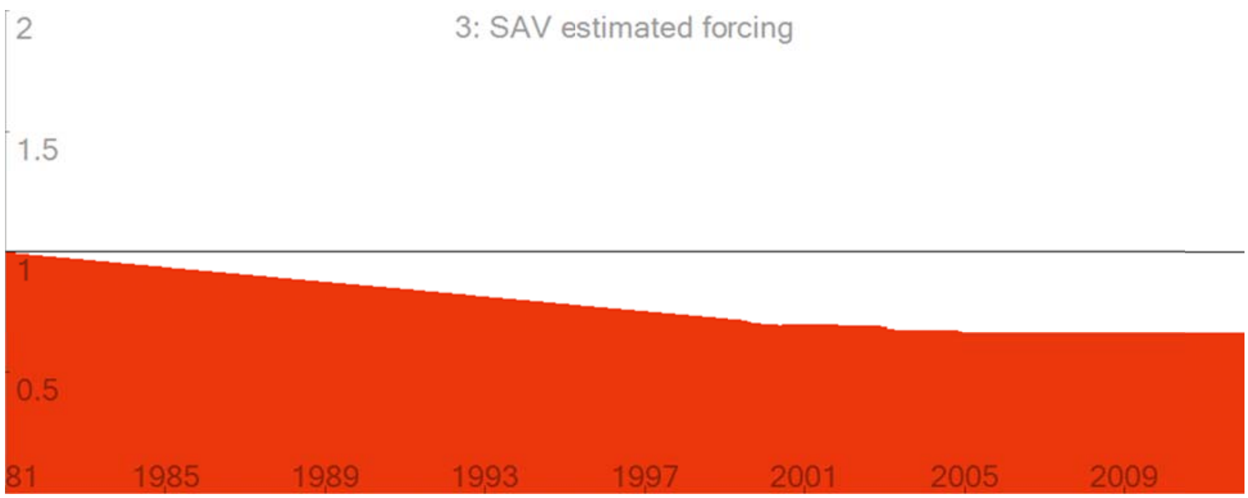


Figure 4: Forcing functions for the EcoSim model run; benthic algae (top), and SAV (bottom).





Nutrient-Phytoplankton-Zooplankton Model

The details and results of the full NPZ model are part of Kevin Crum's Master's thesis "Modeling plankton in a human-impacted estuary: Copepod- vs. jellyfish-dominated Communities", which was approved and accepted by Rutgers University. This thesis formed the basis of a manuscript entitled "Model-data comparisons reveal influence of jellyfish interactions on plankton community dynamics." The manuscript was provided to NJDEP for review on May 12, 2014 and was subsequently submitted to Marine Ecology Progress Series (MEPS), where it was published in December 2014. The published manuscript is attached here as Appendix 5.

Objective 3 - Write and test the program to dynamically link the NPZ and EwE models

The original plan as laid out in the proposal anticipated linking the EwE model to the NPZ model in order to more completely capture the interactions between the lower trophic levels (phytoplankton and zooplankton) and the upper level consumers. This linkage proved to be especially problematic given the different time steps at which the models operate and the internal architecture of the models. While assessing the best way to link the models we were contacted by the USGS Joint Ecological Modeling (USGS-JEM) group to see if they could provide any assistance with data visualization products or model linkages. The USGS received funding to provide assistance to modeling projects within the areas affected by Superstorm Sandy, and they were interested in our project. After a series of emails and conference calls describing our model structure, our needs, and their technical capabilities, we had a meeting March 20-21, 2014 in New Brunswick to outline a plan and timeline for collaboration. At this meeting it was agreed that the USGS-JEM group would build a suite of new visualization tools within the existing EwE software package. Furthermore, the USGS-JEM group will assist in development of a linkage that takes the phytoplankton biomass and production/biomass rates generated by the WASP water quality model being developed by the USGS New Jersey Water Science Center for the Department and pass that information into the EwE model. This model coupling will allow the upper trophic levels of the EwE model to be responsive to changes in nutrients, temperature, or other environmental or management factors that primarily act on lower trophic levels and may not be suitably modeled in EwE. There were some delays in the construction of the WASP model, and therefore this link between WASP and EwE is one of our Year 3 project goals.

However, we have been working with the JEM group to make sure that we will have comparable model groups for when we begin model integration.

Objective 3 Revised – Field assessment of otter trawl efficiency

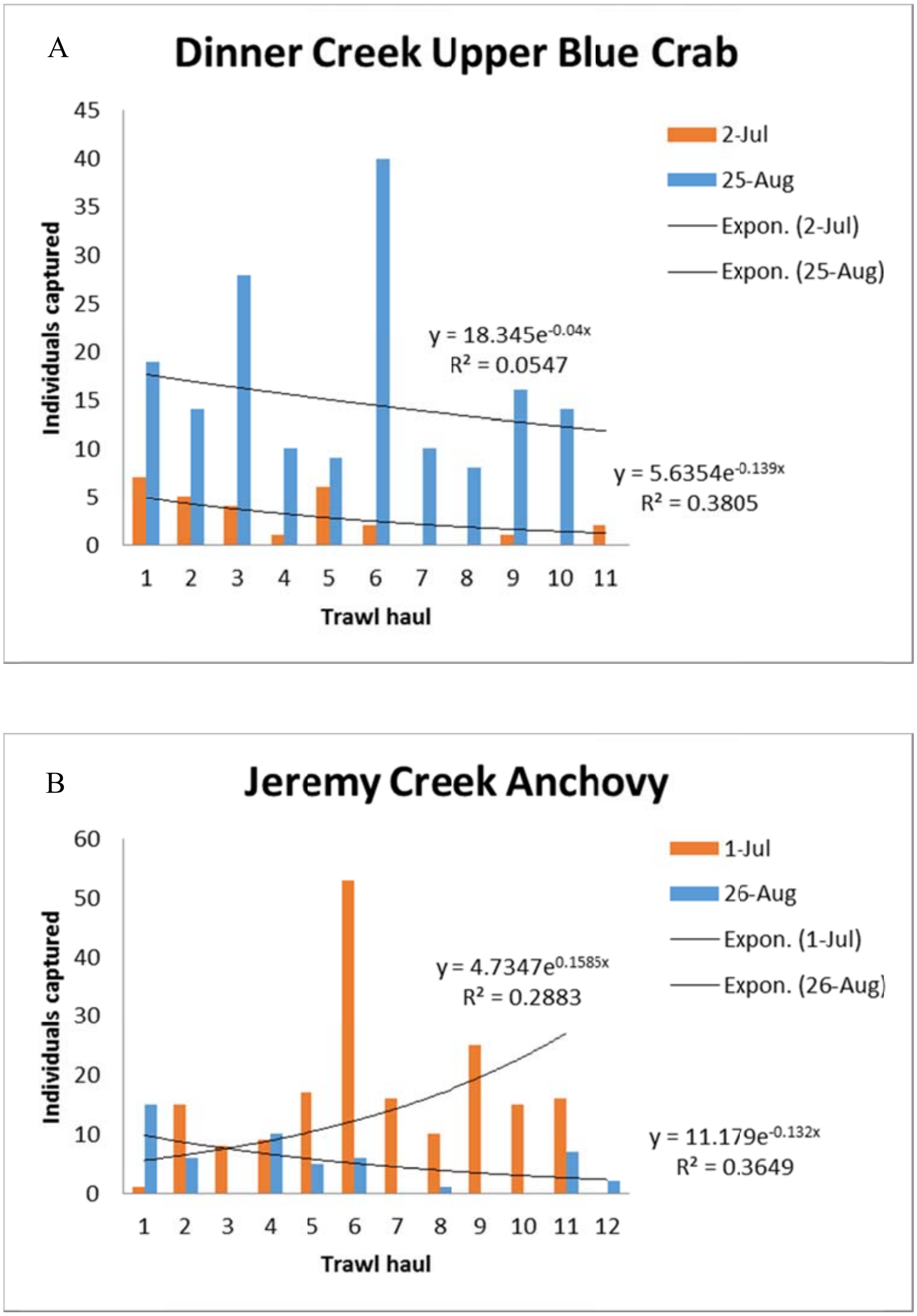
Funds originally allocated for Objective 3 (above) were reallocated toward a field assessment of otter trawl efficiency with the approval of the NJDEP (email from Tom Belton to Olaf Jensen on May 27, 2014).

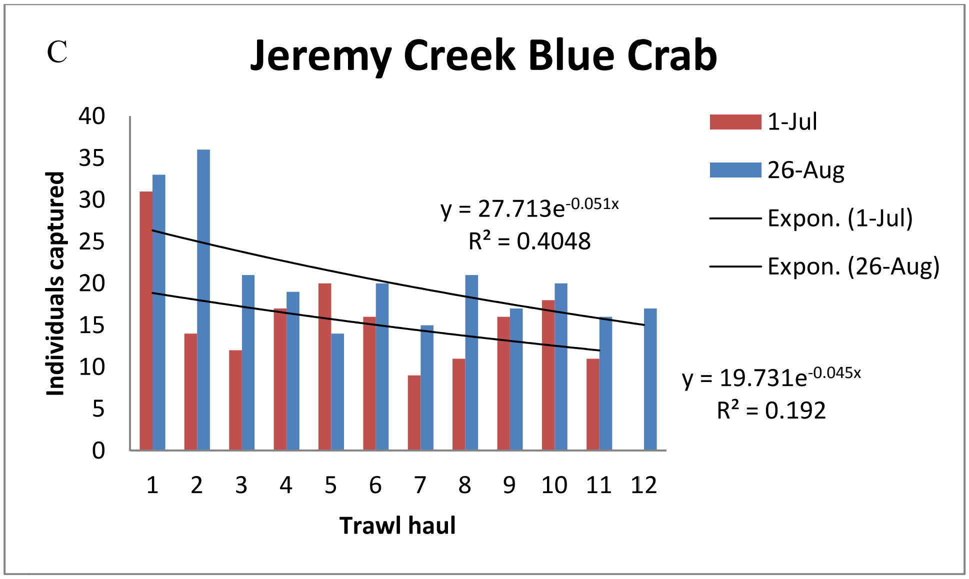
One of the major challenges with development of the EwE ecosystem model is estimating absolute biomass for each of the different trophic groups in Barnegat Bay that are represented in the model. No sampling gear is 100% efficient. That is, all sampling gears capture less than 100% of the organisms encountered. Therefore it is inaccurate to simply estimate biomass based on the number of individuals captured divided by the area or volume sampled. In particular, much of the data on fish and macroinvertebrate abundance used in the EwE model comes from the Rutgers University Marine Field Station's (RUMFS) otter trawl survey.

We conducted field assessments of otter trawl efficiency in two sampling events – July 1-3 and August 25-27, 2014 – in three tributaries of Little Egg Harbor. Sites were similar in size, temperature, and salinity to many of the marsh creeks in Barnegat Bay, but were more easily sampled from RUMFS. We set block nets (< 5 mm mesh) across the width of the marsh creek at two locations approximately 50 m apart to isolate the sampled reach from ingress or egress of fish and blue crab. The isolated section of the creek was then repeatedly trawled and all fish and blue crab that were captured were identified to species, recorded, and either removed from the isolated section of the creek (fish) or, for crabs, a leg was clipped at the terminal segment to mark the individual as previously captured and the crab was returned to the isolated section. Catch for the two taxa captured in sufficient numbers (bay anchovy and blue crab) were plotted for each trawl haul and, where appropriate, an exponential curve was fit to the data to estimate the rate of depletion.

There were four site x species combinations, one for bay anchovy and three for blue crab, for which the exponential model was an adequate representation of the observed data (Figure 5). For the other site x species combinations there were either too few individuals captured or no apparent decline in catch. No decline in catch might occur if the trawl efficiency is very low and the abundance of a given species is high or if the block nets did not prevent immigration into the isolated creek section.

If we compare the catch from the first trawl haul to the total catch expected if the isolated creek section were trawled to depletion, we can estimate the trawl efficiency. Trawl efficiency estimated in this manner for blue crab ranged between 4.2% and 22.2% with an average of 11.7%. Efficiency was not estimated for bay anchovy as there was only a single occasion at a single site in which a clear decline in catch was apparent for this species.


A
Dinner Creek Upper Blue Crab
2-Jul
25-Aug
Expon. (2-Jul)
Expon. (25-Aug)
y = 18.345e-0.04x
R² = 0.0547
y = 5.6354e-0.139x
R² = 0.3805
Individuals captured
Trawl haul
B
Jeremy Creek Anchovy
1-Jul
26-Aug
Expon. (1-Jul)
Expon. (26-Aug)
y = 4.7347e0.1585x
R² = 0.2883
y = 11.179e-0.132x
R² = 0.3649
Individuals captured
Trawl haul



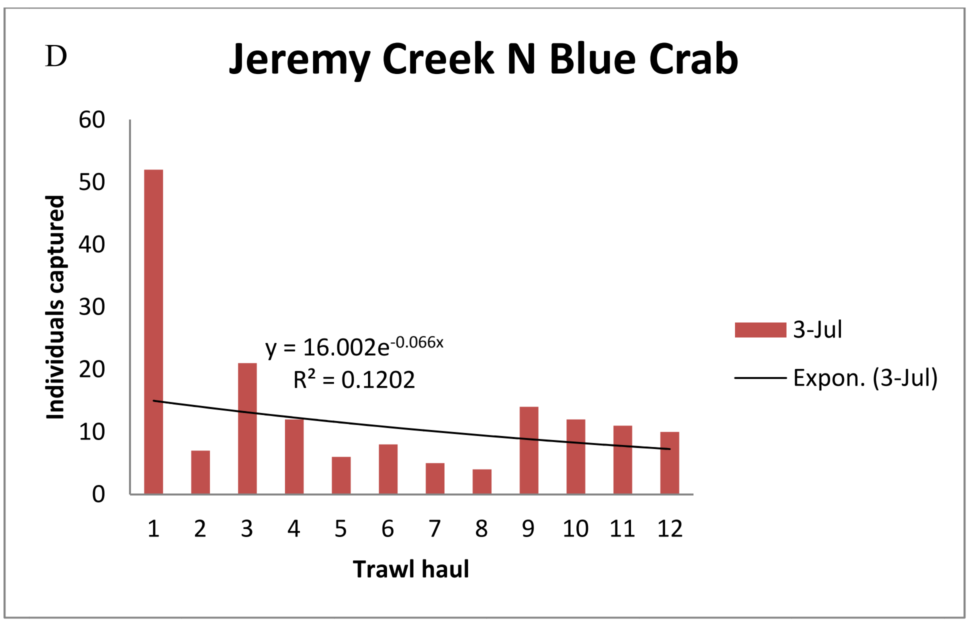


Figure 5. Catch for each trawl haul for blue crab (A, C, and D) and bay anchovy (B) in three locations within tributaries of Little Egg Harbor. Lines represent exponential models fit to the observed catch.

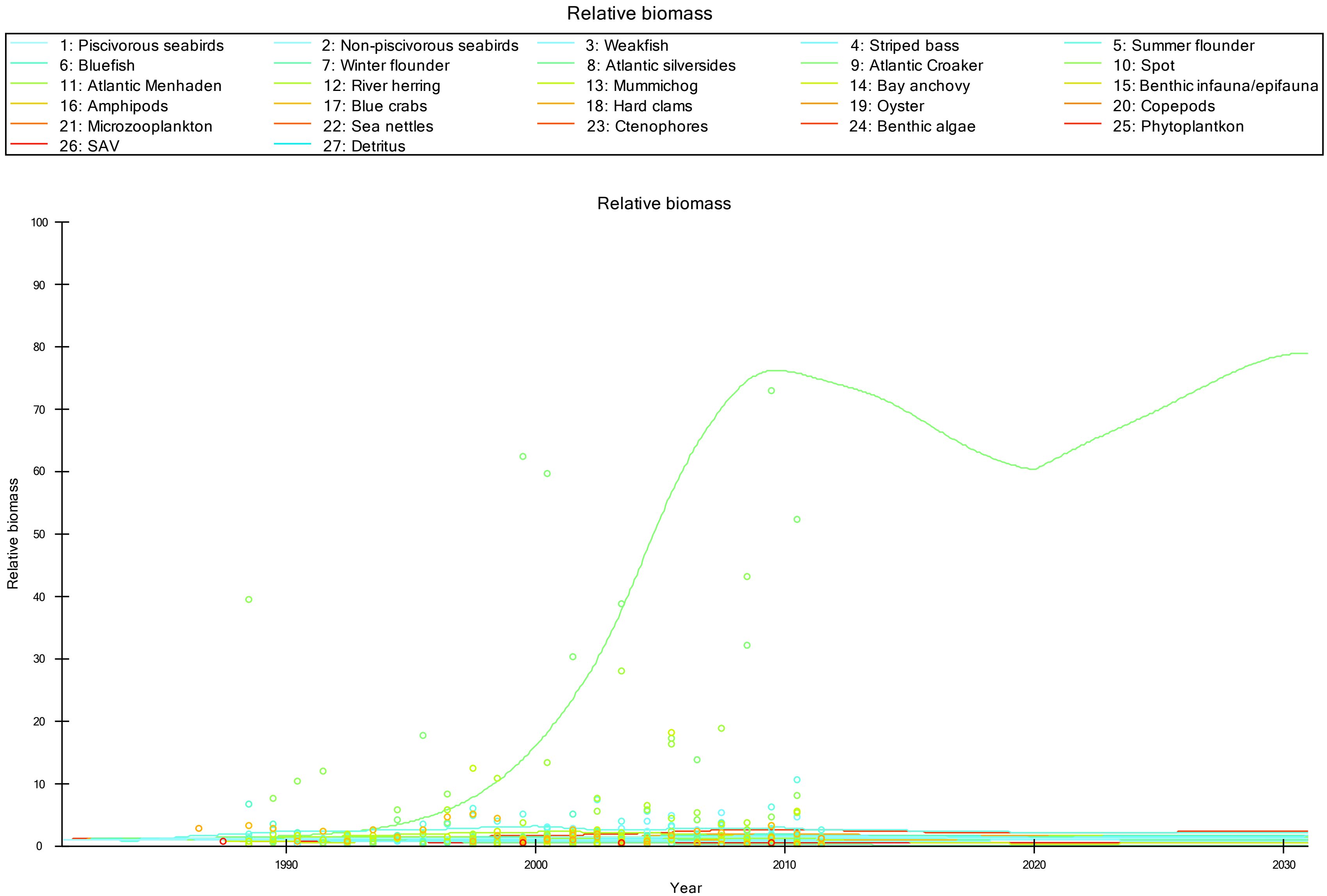
Objective 4 and 5- Develop and run quantitative change scenarios

During the conceptual model development interviews we asked individuals to tell us what variables in their cognitive maps they would increase or decrease the values of in order to effect "positive" change on the bay ecosystem. The responses included changes to both social and ecological components of the system, the impacts of some of which we can model in the EwE, and to a lesser extent in the NPZ, models. We also received input from scientists and managers within the Department regarding what change scenarios they would be most interested in. The following are the results of those changes based on the model as described above.

Scenario 1 – Oyster Creek Nuclear Generating Station (OCNGS) closure

As America's oldest continuously operating nuclear plant, the facility uses a once-through cooling water system, where water is drawn from Forked River, used to cool the plant, and is then returned to Oyster Creek to flow into the bay. The impingement and entrainment of fish, crab, and hard clam larvae, as well as other zooplankters, is well documented. OCNGS functions as a *de facto* fishery, and the removal of biomass from the system is accounted for through catch data used in the EwE model. As part of the Governor's 10-point Plan, the Oyster Creek Nuclear Generating Station (OCNGS) will cease power generation by 2020. To model this scenario we reduced the "catch" of the plant from the "catch" at full operating capacity to 4% of the full operating capacity beginning in 2020, based on the percent reduction in intake water that is planned. The time series data was amended so that the 2011 values for the forced effort series were used for 2012-2030, with the previously noted exception of OCNGS effort. The benthic macroalage, and SAV forcing was set to the 2011 level for the remainder of the simulation. Under those model parameters the relative biomass of most of the groups remains relatively flat or continues along a previous trend, though croaker appears to increase following the plant reduction (Figure 6). If the forced effort data for 2012-2030 are assumed to be the average of the 1981-2011 data the results are similar, though the croaker rebound is dampened slightly.

Figure 6: Model predictions assuming a 96% reduction of OCNGS water uptake from the 1981 value beginning in 2020.



Scenario 2 – Changes to blue crab management strategy

Blue crab are the target of Barnegat Bay's largest commercial fishery, and are currently managed based on a mix of sex and size limits and seasonal closures (NJAC7E:25 and 25A). We modeled the effects of increasing the commercial dredge harvest to 88 metric tons (twice the 1995-2011 average of 44 MT.) and of decreasing the commercial dredge harvest to 22 MT (one-half the ten year average) from 2012 to 2030, while keeping the commercial pot fishery and recreation fisheries at their 1995-2011 averages and the other effort series at their 2011 values. Doubling or halving the commercial dredge had little effect on crab biomass (Figure 7). We also modeled the effects of doubling the commercial pot fishery over the 1995-2011 average of 210 MT to 420 MT and of halving it to 105 MT. Reducing or increasing the landings in the commercial pot fishery had little effect on crab biomass (Figure 8). Even with the commercial pot fishery effort doubled, total catches never exceeded $3MT/km^2$, while biomass was predicted to remain steady near $7.5MT/km^2$. That is, even a doubling of effort in the commercial pot fishery results in a catch that is too small to have a major impact on the blue crab biomass given estimates of unfished biomass and productivity. We are re-examining estimates of blue crab biomass and productivity using a stock assessment model applied to blue crab landings data.

Scenario 3 – Changes to hard clam management strategy

Hard clams were historically one of the most important commercial fisheries in the Bay, but landings have declined dramatically over the past several decades. We will model the effects of limiting the commercial harvest to 25,000 lbs. (the average of the available landings during the 2000's) during the prediction period (2012-2030) and of closing the fishery entirely for a period of ten years (2012-2022) and then returning to the 25,000 lbs limit. Limiting the commercial harvest to 25,000 lbs. appears to have no effect on hard clam biomass as it fluctuates around 40 t/km2 subsequent to 2011 (Figure 9, left panel). This appears to be primarily driven by natural mortality, which displays a similar pattern. A ten-year moratorium on commercial landings showed identical results (Figure 9, right panel). Both the catch and fishing mortality after 2000 are such a small percentage of the total biomass and total mortality, respectively, that harvest controls have little effect on the population. The large caveats here are that hard clam landings are not recorded by the NJDEP or NMFS, and thus the landing data we obtained appear to be estimates with potentially large uncertainty. Furthermore, the relative paucity of data on hard clams in Barnegat Bay over time made fitting the model particularly difficult for this species.

Figure 7: Changes to the biomass (t/km^2) of blue crab (*Callinectes sapidus*) post 2011 following a doubling of the average dredge fishery effort from 1995-2011 (left panel) and a halving of the effort (right panel).

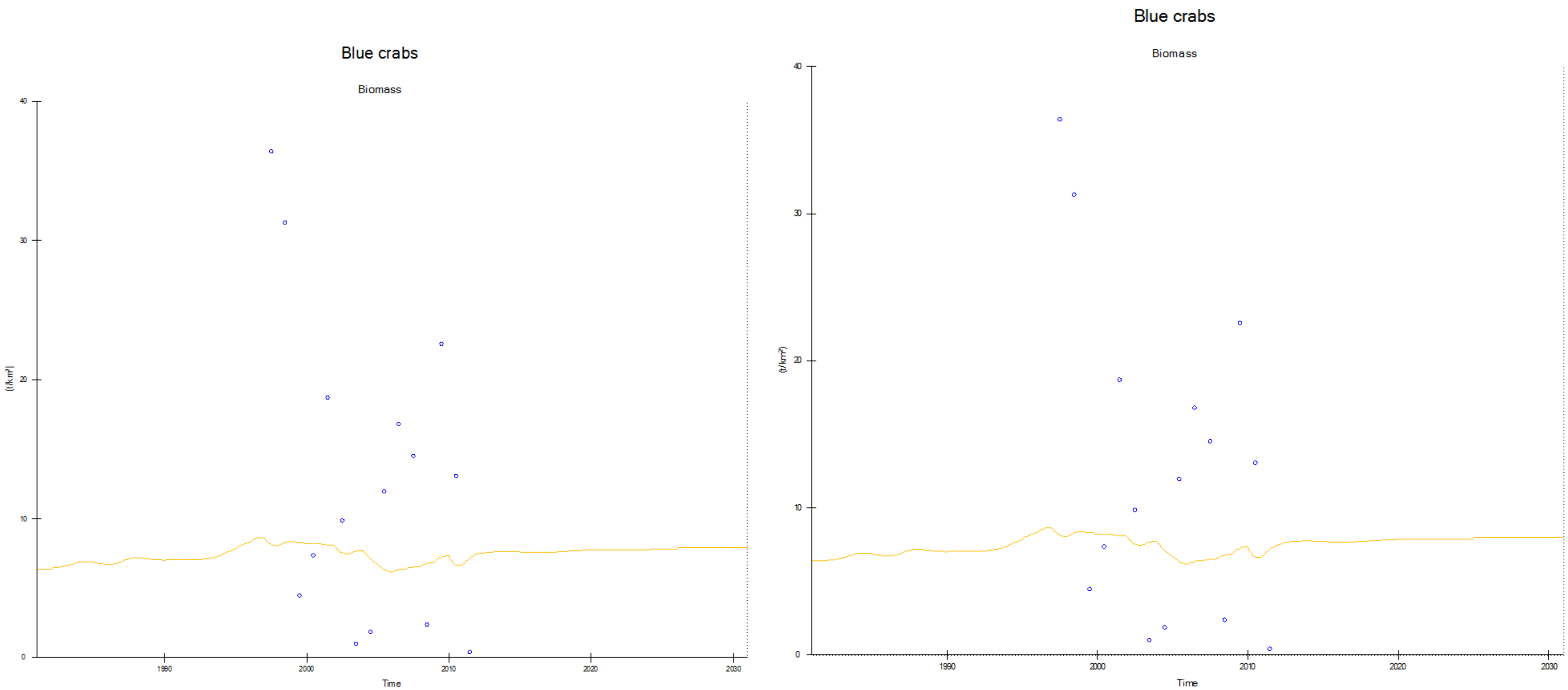


Figure 8: Changes to the biomass (t/km^2) of blue crab (*Callinectes sapidus*) post 2011 following a doubling of the average commercial pot fishery effort from 1995-2011 (left panel) and a halving of the effort (right panel).

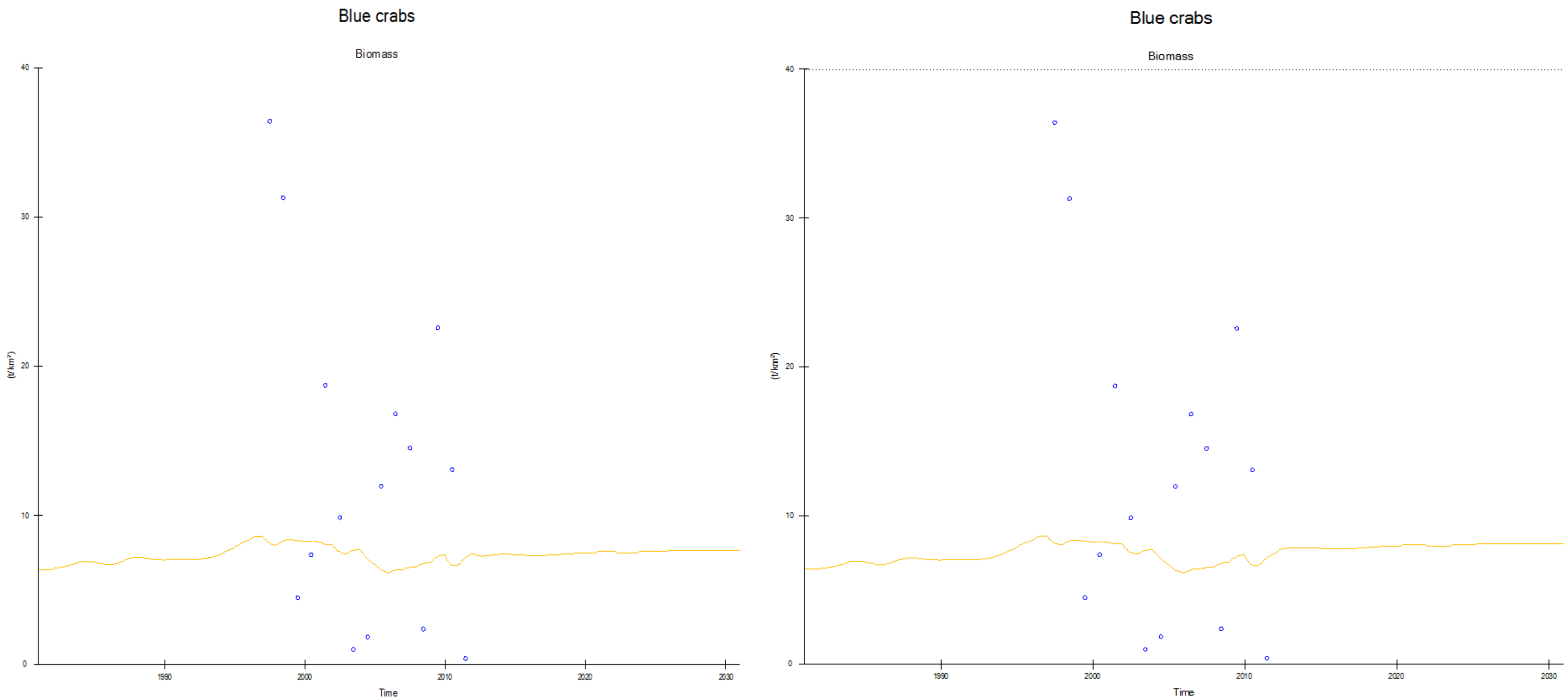
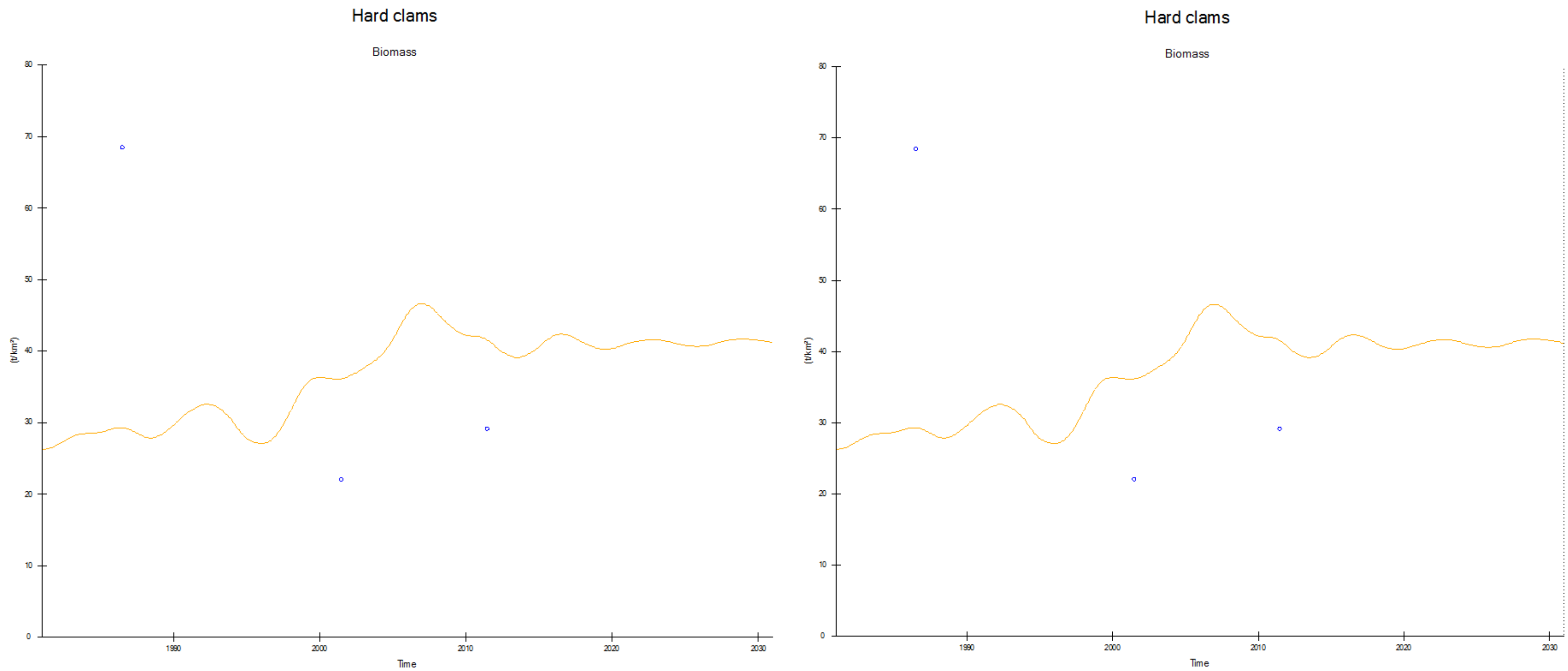


Figure 9: Changes to the biomass (t/km^2) of hard clam (*Mercenaria mercenaria*) post 2011 following a harvest restriction of 25,000 lbs. (left panel) and a closure of the fishery (2012-2012) followed by a limited harvest (right panel).



Scenario 4 – Nutrient input reduction

The Barnegat Bay has been described as a highly eutrophic estuarine system (e.g., Kennish et al. 2007), and the focus of recent legislation (NJ Fertilizer Act, P.L. 2010 Chapter 112; NJ Soil Restoration Act, P.L. 2010 Chapter 113) and restoration efforts (NJ Stormwater Act, P.L. 2010 Chapter 114; Clean Water Act Section 319 projects) in New Jersey has been to reduce the amount of nitrogen being delivered into the system. As no target reductions have been set at this time, we propose to model the effects of reducing nitrogen inputs by 5% and 15%. The effects of these reductions will be felt most directly on phytoplankton and zooplankton biomass, and can be most appropriately modeled in the WASP model that is currently under production. Once the linkage between the WASP and EwE model is constructed we will be able to pass the changes along to the upper trophic levels.

References

Anderson, D.R., 1975. Population ecology of the mallard, V: Temporal and geographic estimates of survival, recovery, and harvest rates. U.S. Fish and Wildl. Serv. Resour. Publ., 25:110.

ASMFC. 2003 Atlantic striped bass advisory report. ASMFC Striped Bass Technical Committee Report 2003-03, Atlantic States Marine Fisheries Commission, Washington, D.C.

Baird, D. and Ulanowicz, R.E., 1989. The seasonal dynamics of the Chesapeake Bay ecosystem. Ecological Monographs, 59:329-364.

Bougon, M., Weick, K., Binkhorst, D., 1977. Cognition in organizations: an analysis of the Utrecht Jazz Orchestra. Admin. Sci. Quart. 22, 606–639.

Bricelj, V. M., J.N. Kraeuter, and G. Flimin. 2013. Status and Trends of Hard Clam, *Mercenaria mercenaria*, Shellfish Populations in Barnegat Bay, New Jersey. Barnegat Bay Partnership Technical Report. Toms River, Barnegat Bay Partnership**:** 143.

Christensen, V. and Walters, C.J., 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecol. Model., 172:109-139.

Christensen, Villy, and Alasdair Beattie, Claire Buchanan, Hongguang Ma, Steven J. D. Martell, Robert J. Latour, Dave Preikshot, Madeline B. Sigrist, James H. Uphoff, Carl J. Walters, Robert J. Wood, and Howard Townsend. 2009. Fisheries Ecosystem Model of the Chesapeake Bay: Methodology, Parameterization, and Model Explanation. U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-F/SPO-106, 146 p.

Christensen, V., Walters, C.J. and Pauly, D., 2005. Ecopath with Ecosim: a User's Guide, November 2005 Edition, Fisheries Centre, University of British Columbia, Vancouver, Canada.

Carley, K., Palmquist, M., 1992. Extracting, representing, and analyzing mental models. Social Forces 70, 601–636.

Eden, C., Ackerman, F., Cropper, S., 1992. The analysis of cause maps. J. Manage. Stud. 29, 309–323

Frisk, M.G., T.J. Miller, R.J. Latour, and S. Martell. 2006. An ecosystem model of Delaware Bay.

Froese, R. and Pauly, D., 2004. FishBase, World Wide Web electronic publication, www.fishbase.org,version (03/2013).

Fuchs HL, Franks PJS (2010). Plankton community properties determined by nutrients and size selective feeding. Marine Ecology Progress Series, 413: 1-15.

Gray, S., A. Chan, D. Clark, R. Jordan. 2011. Modeling the integration of stakeholder knowledge in social–ecological decision-making: Benefits and limitations to knowledge diversity. Ecol. Model. doi:10.1016/j.ecolmodel.2011.09.011

Hage, P., Harary, F., 1983. Structural Models in Anthropology. Oxford University Press, New York.

Harary, F., Norman, R.Z., Cartwright, D., 1965. Structural Models: An Introduction to the Theory of Directed Graphs. John Wiley & Sons, New York.

Hobbs, B.F., Ludsin, S.A., Knight, R.L., Ryan, P.A., Biberhofer, J., Ciborowski, J.J.H., 2002. Fuzzy cognitive mapping as a tool to define management objectives for complex ecosystems. Ecol. Appl. 12, 1548–1565.

Houde, E.D. and Zastrow, C.E., 1991. Bay anchovy (*Anchoa mitchilli*). In: S.L. Funderburk, J.A. Mihursky, S.J. Jordon and D. Riley (Editor), Habitat requirements for Chesapeake Bay living resources. 2nd edition. Chesapeake Bay Program Office, U.S. Environmental Protection Agency, Annapolis, Md., pp. 8:1-14.

Hoenig, J. M. 1983. Empirical Use of Longevity Data to Estimate Mortality-Rates. Fishery Bulletin **81**:898-903.

ICES, 2000. Report of the working group on seabird ecology, ICES CM 2000/C:04

Jørgensen, L.A., Jørgensen, S.E. and Nielsen, S.N., 2000. ECOTOX: Ecological Modelling and Ecotoxicology. Elsevier Science B.V., Amsterdam.00

Kahn, D. M. 2003. Stock assessment of Delaware Bay blue crab (Callinectes sapidus) for 2003. Div. Fish Wild., Dover, DE.

Kahn, D. M., and Helser T. E. 2005. Abundance dynamics and mortality rates of the Delaware Bay stock of blue crabs, Callinectes sapidus. Journal of Shellfish Research **24**:269-284.

Kennish, M.J. 2001. The Scientific Characterization of the Barnegat Bay – Little Egg Harbor Estuary and Watershed. Jacques Cousteau National Estuarine Research Reserve Contribution #100-5-01.

Kennish MJ (2001a). Physical description of the Barnegat Bay—Little Egg Harbor estuarine system. Journal of Coastal Research, SI(32): 13-27.

Kennish, M.J., S.B. Bricker, W.C. Dennison, P.M. Glibert, R.J. Livingston, K.A. Moore, R.T. Noble, H.W. Paerl, J.M. Ramstack, S. Seitzinger, D.A. Tomasko, and I. Valiela. 2007. Barnegat Bay–Little Egg Harbor Estuary: case study of a highly eutrophic coastal bay system. Ecological Applications 17: S3–S16.

Kennish, M.J., B.M. Ferting, G.P. Sakowicz. 2013. In situ Surveys of Seagrass Habitat in the Northern Segment of the Barnegat Bay - Little Egg Harbor Estuary: Eutrophication Assessment. Barnegat Bay Partnership Technical Report. 43p.

Kim, H.S., Lee, K.C., 1998. Fuzzy implications of fuzzy cognitive map with emphasis on fuzzy causal relationships and fuzzy partially causal relationship. Fuzzy Sets Syst. 97, 303–313.

Kosko, B., 1986. Fuzzy Cognitive Maps. Int. J. Man–Machine Stud. 24, 65–74.

Lathrop, R. G. , R.M. Styles, S. P. Seitzinger, J.A. Bognar. 2001. Use of GIS Mapping and Modeling Approaches to Examine the Spatial Distribution of Seagrasses in Barnegat Bay, New Jersey. Estuaries 24(6A): 904-916.

Lowerre-barbieri, S. K., Chittenden M. E., and Barbieri L. R. 1995. Age and Growth of Weakfish, Cynoscion Regalis, in the Chesapeake Bay-Region with a Discussion of Historical Changes in Maximum Size. Fishery Bulletin **93**:643-656.

Luo, J. and Brandt, S.B., 1993. Bay anchovy, Anchoa mitchilli, production and consumption in mid-Chesapeake Bay based on a bioenergetics model and acoustic measurement of fish abundance. Marine Ecology Progress Series, 98:223-236.

Macro International Inc. 2008. New Jersey Blue Crab Recreational Fishery Survey 2007 Final Report.

Matishov, G.G. and Denisov, V.V., 1999. Ecosystems and biological resources of Russian European seas at the turn of the 21st century, Murmansk Marine Biological Institute, Murmansk

Moser FC (1997). Sources and sinks of nitrogen and trace metals, and benthic macrofauna assembleges in Barnegay Bay, New Jersey. PhD Dissertation. Rutgers University, New Brunswick, New Jersey, USA.

Nemerson, D. M., and Able K. W. 2004. Spatial patterns in diet and distribution of juveniles of four fish species in Delaware Bay marsh creeks: factors influencing fish abundance. Marine Ecology-Progress Series **276**:249-262.

Olsen PS, Mahoney JB (2001). Phytoplankton in the Barnegat Bay-Little Egg Harbor estuarine system: Species composition and picoplankton bloom development. Journal of Coastal Research, SI(32): 115-143.

Oshima, Y., Kishi, M.J. and Sugimoto, T., 1999. Evaluation of the nutrient budget in a seagrass bed. Ecol Model, 115:19-33.

Özesmi, U., Özesmi, S., 2003. A participatory approach to ecosystem conservation: fuzzy cognitive maps and stakeholder group analysis in Uluabat Lake, Turkey. Environ. Manage. 31 (4), 518–531.

Özesmi, U., Özesmi, S., 2004. Ecological models based on people's knowledge: a multi-step fuzzy cognitive mapping approach. Ecol Model. 176:43-64.

Palomares, M. L. D. 1991. La consummation de nourriture chez les poissons: étude comparative, mise au point d'un modèle prédictif et application à l'étude des réseaux trophiques. Thèse de Doctorat, Institut National Polytechnique de Toulouse:211.

Palomares, M.L.D. and Pauly, D., 1998. Predicting food consumption of fish populations as functions of mortality, food type, morphometrics, temperature and salinity. Mar. Freshwat. Res., 49:447-453.

Park, G.S. and Marshall, H.G., 2000. The trophic contributions of rotifers in tidal freshwater and estuarine environments. Estuarine, Coastal and Shelf Science, 51:729-742.

Pauly, D. 1989. Food consumption by tropical and temperate fish populations: some generalizations. J. Fish Biol. **35(Suppl. A)**:11-20

Piner, K. R., and Jones C. M. 2004. Age, growth and the potential for growth overfishing of spot (Leiostomus xanthurus) from the Chesapeake Bay, eastern USA. Marine and Freshwater Research **55**:553-560.

Preikshot, D., 2007. The influence of geographic scale, climate and trophic dynamics upon North Pacific oceanic ecosystem models. Ph.D. , University of British Columbia, Vancouver

Randall, R.G. and Minns, C.K., 2000. Use of fish production per unit biomass ratios for measuring the productive capacity of fish habitats. Canadian Journal of Fisheries and Aquatic Sciences, 57:1657-1667.

Ross, S. W. 1988. Age, growth, and mortality of Atlantic croaker in North Carolina, with comments on population dynamics. Trans. Am. Fish. Soc. **117**:461-473.

Sellner, K.G., Fisher, N., Hager, C.H., Walter , J.F. and Latour, R.J., 2001. Ecopath with Ecosim Workshop, Patuxent Wildlife Center, October 22-24, 2001, Chesapeake Research Consortium, Edgewater MD

Shushkina, E.A., Musaeva, E.I., Anokhina, L.L. and Lukasheva, T.A., 2000. The role of gelatinous macroplankton, jellyfish *Aurelia*, and Ctenophores *Mnemiopsis* and *Beroe* in the planktonic communities of the Black Sea. Russian Academy of Sciences. Oceanology, 40:809-816.

Sissenwine, M., 1987. Chapter 31. Fish and squid production. In: R.H. Backus and D.W. Bourne (Editor), Georges Bank. MIT Press, Cambridge, Mass., pp. 347-350.

Smith, D.R., Burnham, K.P., Kahn, D.M., He, X. and Goshorn, C.J., 2000. Bias in survival estimates from tag-recovery models where catch-and-release is common, with an example from Atlantic striped bass. Canadian Journal of Fisheries and Aquatic Sciences, 57:886-997

Sugihara, T., C. Yearsley, J.B. Durand, N.P. Psuty. 1979. Comparison of Natural and Altered Estuarine Systems. Center for Coastal and Environmental Studies, Rutgers – The State University of New Jersey. CCES Publication NJ/RU – DEP-11-9-79.

Tomasko, D. A., C. J. Dawes, M.O. Hall. 1996. "The effects of anthropogenic nutrient enrichment on turtle grass (Thalassia testudinum) in Sarasota Bay, Florida." Estuaries **19**(2B): 448-456.

Appendix 1 – Ecopath Parameter Derivations

Fish

Atlantic Croaker

Q/B - Estimates of consumption to biomass ratio was calculated in FishBase as 4.2 year^{-1}, assuming an annual temperature of the Barnegat Bay of T = 15 oC, aspect ratio = 1.32, W*inf* = 815.3, and carnivorous feeding.

P/B - An annual total mortality for the Chesapeake Bay Atlantic croaker stock was estimated to be 55 to 60% per year (Austin *et al*., 2003). Using the higher end as a conservative mortality estimate yields a P/B = 0.916 year^{-1}.

Biomass – An EE value of 0.90 was used and EwE estimated the biomass. Croaker were rarely identified in the Sugihara et. al (1979) study and thus the Delaware Bay and Chesapeake models likely overestimate the biomass present here.

Diet – The diet data is based on the general diet found in the Delaware Bay model, which is a composite of the Nemerson and Able (1994) study.

Atlantic Menhaden

Q/B – A value of 31.42 year^{-1} taken from Palomares and Pauly (1998).

P/B – As there was no commercial fishery for menhaden in Barnegat Bay and only a limited bait fishery, total mortality was set equal to natural mortality, which is estimated at 0.50 year^{-1} (MSVPA-X averaged across all ages and 1982-2008; in 2010 Stock Assessment Table 2.13).

Biomass – Biomass was calculated by EwE setting the EE to 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Atlantic Silverside

Q/B – The consumption ratio for silversides of 4.0 year^{-1} was determined by setting a production/consumption ratio of 0.2 (Christensen et al.).

P/B – Total mortality for littoral forage fish was estimated by local experts at a Chesapeake Bay Ecopath Workshop (Sellner *et al*., 2001) to be 0.8 year^{-1}.

Biomass - The biomass for the group was estimated by setting ecotrophic efficiency to 0.95. While baywide biomass was not determined by Vougliotis et al (1987), they suggested it should be comparable, if not great than what they determined for bay anchovy, given Atlantic silverside was numerically dominant.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Bay Anchovy

Q/B - Assuming habitat temperature of 15 ^{o}C, W∞ = 20 (g), an aspect ratio of 1.32, and carnivorous diet, the consumption to biomass ratio is calculated by Fishbase to be 9.7 year^{-1}.

P/B – Christensen et al used an initial P/B of 3.0 year^{-1} for the Chesapeake Bay model based on a 95% annual mortality rate reported by Luo and Brandt (1993), while Frisk et al. (2006) estimated a P/B of 2.19 year^{-1} from catch curve analysis on adults in Delaware Bay. We elected to use the higher rate.

Biomass – Vougliotis et al (1987) estimated biomass for 1976 to range from 0.83 to 4.83 g/m^2. In the same study the catch per unit effort for 1981 was comparable to that for 1976, and thus the biomass range should be similar. Given the ubiquity of the species within the Barnegat Bay, I chose to use $4.83 g/m^2$ for an initial biomass.

Diet - Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Bluefish

Q/B - Assuming habitat temperature of 15 $_oC$, W_{max} = 16,962.1 (g), carnivorous feeding, and an aspect ratio of 2.55, the resulting consumption to biomass ratio is 3.1 $year^{-1}$.

P/B – Production/biomass was determined as 0.52 $year^{-1}$ based on an M = 0.25 $year^{-1}$ (Christensen et al) and an estimate of F = 0.27 $year^{-1}$ for 1982 from the 41^{st} Stock Assessment Workshop (2005) for Bluefish (Figure B2).

Biomass – Biomass was calculated by EwE setting the EE to 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study averaged for all size classes.

Mummichog

Q/B – A Q/B of 3.65 $year^{-1}$ was used (Pauly1989).

P/B – We opted to utilize a P/B of 1.2 $year^{-1}$ as given in Frisk et al (2006) from “best professional judgement” compared to Valiela 0.287 $year^{-1}$ (1977 mortality tables) or Christensen et al’s 0.8 $year^{-1}$.

Biomass- The biomass for the group was estimated by setting ecotrophic efficiency to 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

River herring

Q/B – We used a Q/B = 8.4 $year^{-1}$, which is the average of Pauly (1989; 8.63 at temperature = 10C) and Palomares (1991; 8.23 at temperature= 20C).

P/B - Total mortality for this group was based on the P/B of 0.75 $year^{-1}$ for alewife in Randall and Minns (2000).

Biomass – Biomass was estimated by EcoPath assuming that the ecotrophic efficiency of these species in the Bay was 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Spot

Q/B – The consumption biomass ratio was estimated as 6.2 $year^{-1}$ using the model in Fishbase.org and a habitat temperature of 15 $_0C$, W_∞ = 190g (Piner and Jones, 2004) and an aspect ratio of 1.39 (Christensen et al).

P/B - Hoenig’s method estimated an M = 0.9 $year^{-1}$ given a maximum age of 5 (Piner and Jones, 2004). This is consistent with the Z used in the Delaware Bay model.

Biomass – Biomass was estimated by the software using an EE value of 0.90, which was taken from the Chesapeake Bay model.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Striped bass

Q/B - Based on empirical relationship provided by Fishbase.org and assuming an aspect ratio of 2.31 (Chesapeake Bay Ecopath Model), temperature T = 15 ^{0}C, and W_∞ = 46.6 kg (Funderbunk et al 1991), the estimated consumption ratio was 2.4 year^{-1}.

P/B – The 1981 ASMFC FMP suggest an M=.15 and an F=.3 for the coastwide stock. Given the reduced fishing mortality in the Barnegat Bay, an F=.25 is appropriate leading to a P/B of 0.4 year^{-1}. This is equal to the Chesapeake model for resident bass (1-7 years old), though their YOY P/B = 1.8 year^{-1}.

Biomass – The biomass was estimated by EcoPath based on an EE of 0.90.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study and was averaged across all size classes.

Summer Flounder

Q/B- Assuming an aspect ratio of 1.32, W_{max} = 12kg (Frisk et al 2006), carnivorous feeding, and habitat temperature of 15 ^{o}C, the consumption to biomass ratio is = 2.6 year^{-1}.

P/B- The Chesapeake Bay and Delaware Bay models utilized P/B=0.52 year^{-1} based on the 2002 NEFSC determination of M=0.2 and F ranging between 0.24 and 0.32.

Biomass – Biomass was estimated by the software using an EE value of 0.95, which is in-line with that used in the Chesapeake Bay model.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Weakfish

Q/B - Using Fishbase, consumption to biomass was estimated = 3.0 year^{-1}, assuming average habitat temperature of 15 ^{o}C, aspect ratio of 1.32, maximum weight W_∞ = 6,190g (Lowerre-Barbieri et al., 1995) and carnivorous feeding habitats.

P/B –Total mortality of Z = 0.26 year^{-1} was estimated using Hoenig's method (1983) assuming a longevity of 17 years (Lowerre-Barbieri et al., 1995). This is in-line with an estimated M of .25 year^{-1} as used for stock assessment purposes (Smith *et al.*,2000). Given the low rate of fishing in Barnegat Bay, Hoenig's estimation of Z seem reasonable.

Biomass – Biomass was estimated by the software using an EE value of 0.90.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study and averaged across all size classes.

Winter Flounder

Q/B - The estimated consumption ratio of 3.4 year^{-1} was derived using the empirical equation in FishBase (Froese and Pauly, 2004), and was calculated assuming that T = 15 °C, W_{inf} = 3,600 g (Fishbase), an aspect ratio of 1.32, and a carnivorous diet.

P/B – The 2011 Southern New England/Mid-Atlantic stock assessment updated natural mortality (M) to 0.30 year^{-1} for all ages and all years. Fishing mortality for ages 4-6 was determined as 0.61 year^{-1} for 1981. If one assumes only natural mortality for ages 0-3 and then F+M for ages 4-6, total mortality (Z) is 0.52 averaged across all ages.

Biomass – Biomass was estimated by the software using an EE value of 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Piscivorous seabirds

Q/B - The consumption ratio estimate of 120 $year^{-1}$ was from data for the piscivorous seabirds group in Preikshot (2007).

P/B - A total mortality estimate for piscivorous seabirds of 0.163 $year^{-1}$ was based on survival rate values of 85-90% for cormorants and 80-93% for alcids in the northeast Atlantic (ICES, 2000).

Biomass - The biomass estimate for piscivorous seabirds of 0.25 $t \cdot km^{-2}$ is a reduction of the Chesapeake Bay model estimate (Sellner *et al.*, 2001).

Diet compositions - The diet composition for piscivorous seabirds was taken from the Chesapeake Bay model and was modified by reducing predation on menhaden and increasing imports based on the large number of migratory seabirds.

Non-Piscivorous seabirds

Q/B - The consumption ratio estimate of 120 $year^{-1}$ was from data for the non-piscivorous seabirds group in Preikshot (2007).

P/B - A total mortality estimate for non-piscivorous seabirds of 0.51 $year^{-1}$ was taken from the Chesapeake model and was based on annual mortality rate of 37% for mallard males and 44% females (Anderson, 1975).

Biomass - The biomass estimate for non-piscivorous seabirds of 0.121 $t \cdot km^{-2}$ was taken from the Chesapeake Bay model and was based on advice provided in a Chesapeake Ecopath Workshop (Sellner *et al.*, 2001).

Diet compositions - The diet composition for non-piscivorous seabirds was taken from the Chesapeake Bay model.

INVERTEBRATES

Blue crabs

Q/B- The consumption ratio of 4.0 $year^{-1}$was taken from the Chesapeake Bay model.

P/B – The Delaware Bay model utilized a P/B= 1.21 $year^{-1}$. This was based on a stock assessment for Delaware Bay that used a natural morality of M = 0.8 $year^{-1}$ assuming a lifespan of 4 years (Kahn, 2003) and fishing mortality on total stock (recruits and post recruits) was F = 0.41 $year^{-1}$ (2000-2002).

Biomass – Biomass was estimated using an ecotrophic efficiency of 0.95.

Diet – Diet taken from Chesapeake Bay model, averaged across stanzas.

Hard Clams

Q/B - The consumption ratio was estimated to be 5.1 $year^{-1}$ assuming a P/Q = 0.20 (Chesapeak Bay Model)

P/B - A total production/biomass ratio of 1.681 $year^{-1}$ was calculated using Brey's Multi-parameter P/B model (Brey). This assumes an average mass of 20 g, water T = 15 °C, non-motile behavior, an average water depth of 1.5 m, and a joules to biomass conversion ratio of 1.28J per mg of wet weight with shell (Brey et al 2010, see conversion worksheet).

Biomass – 26.18 t/km2. This is based on a density of 1,309,233 clams per km^2 (adjusted values for the 1985-1987 surveys, Celestino 2002) and an average mass of 20 g (mean length of 7.46cm, Celestino 2013, length to weight average relationship verified 10/27/13 by JV in supermarket).

Diet – Diet taken from Chesapeake Bay model.

Oyster

Q/B - The Q/B ratio of 2.0 $year^{-1}$ was taken from the adult stanza of the Chesapeake Bay Model.

P/B – A 2009 survey of the restored oyster reef at Good Luck Point determined a mean annual mortality of 47%, or an M=0.63 $year^{-1}$ (Calvo 2010). As oysters in Barnegat Bay are an unfished resource, Z=M=.63 $year^{-1}$.

Biomass – Based on NJDEP experience there does not appear to be a viable oyster set in Barnegat Bay; the known oyster reef is seeded by the NJDEP. In order to keep oysters in the model for future management considerations the biomass was set to 0.001t/km^2 to simulate a very small population.

Diet – Data taken from the Chesapeake model.

Sea Nettles

Q/B – A Q/B of 20 $year^{-1}$ was taken from the Chesapeake Bay model. This value is based on an assumed P/Q of 0.25.

P/B – As reported in the Christensen et al (2006), Matishov and Denisov (1999) estimated a daily growth rate for *Aurelia aurita* of 0.053 at 5 ºC to 0.15 at 16.5 ºC. Sea nettle medusa are present in the Barnegat Bay during the summer months, when waters are typically warmer than 16.5 ºC. As such the P/B for Barnegat Bay was calculated as (0.15*365)/4 ~ 13 year $^{-1}$.

Biomass – A biomass of 1.38 t/km^2 (0.92 under old volume) was calculated using bay-wide survey data from Monmouth University for 2012 and an average wet weight of 56g for individuals between 35mm-144mm. Because there are no reports of sea nettles in Barnegat Bay until the later 1980s -early 1990s this initial population is completely removed via "dummy" fishing fleet, whose effort is reduced over time.

Diet – The sea nettle diet data was taken from the Chesapeake Bay model (no citations given)

Ctenophores

Q/B - Shushkina *et al.* (1989) found that ctenophores in their study had growth rates 1.5 to 2 times greater than true jellyfish. Therefore, the Q/B value for ctenophores was the value for sea nettles multiplied by 1.75, *i.e.*Q/B was 35 $year^{-1}$.

P/B – Shushkina *et al.* (1989) found that ctenophores in their study had growth rates 1.5 to 2 times greater than true jellyfish. Ctenophores tend to be present in Barnegat Bay at cooler temperatures than those of sea nettles, therefore the P/B was calculated as 1.75 times the average estimated daily growth rate of *Aurelia aurita* over the course of 3 months ((((0.053+0.15)/2)*365)/4)*1.75 ~ 16.2 $year^{-1}$.

Biomass – A biomass of 7.86 t/km^2 was calculated using bay-wide survey data collected by Monmouth University during 2012 and an average weight of 3.42g per individual.

Diet - The ctenophore diet data was taken from the Chesapeake Bay model (no citations given)

Benthic infauna/epifauna (shrimp, worms, non-blue claw crabs)

Q/B – A consumption ration of 5.0 $year^{-1}$ was estimated by Ecopath after designating a P/Q ratio of 0.2, as taken from the Chesapeake Bay Model.
P/B – A P/B of 2.0 $year^{-1}$ was taken from the Chesapeake Bay model.
Biomass – Estimated by Ecopath, based on a group ecotrophic efficiency of 0.9 as taken from the Chesapeake Bay model.
Diet – Diet data taken from Chesapeake Bay model.

Amphipods

Q/B – Ecopath estimated a Q/B = 5.0 $year^{-1}$ using a P/Q ratio of 0.2, following the Chesapeake Bay model.
P/B – A P/B of 3.8 $year^{-1}$ was used based on the average P/B of *Ampelisca abdita* at 3 locations within Jamaica Bay (Franz and Tanacredi 1992). *A. abdita* was the most common amphipod found in Barnegat Bay sampling in 2012.
Biomass – The biomass of amphipods was estimated by Ecopath using an EE=0.900. We attempted to utilize the first year of NJDEP Barnegat Bay research program data, which is the only study of amphipod density bay-wide, though it is restricted to summer sampling only. A 1974/1975 study (Haskin and Ray 1979) documented amphipod density throughout the year, but on a limited spatial scale. In the 1974/75 study the average yearly density across all sites was approximately 2.5 times larger than the summer density during the same time period. To estimate amphipod biomass, the average density of the 2012 study was multiplied by 2.5, and the resulting density multiplied by the weight of an average amphipod (0.003g) to reach an estimate of 1.53g/m^2. This empirically determined biomass is approximately one-half of the biomass required to balance the model as found by Ecopath.
Diet – The diet data for this group was taken from the benthic infauna group.

Copepods (Mesozooplankton)

Q/B – A consumption ration of 83.333 year $^{-1}$ was estimated by Ecopath after designating a P/Q ratio of 0.3, as taken from the Chesapeake Bay Model.
P/B – A mortality rate of 25 year $^{-1}$ was taken from the Chesapeake Model, as estimated during the Chesapeake Bay Ecopath Workshop (1989).
Biomass – Copepod biomass was estimated using an ecotrophic efficiency of 0.95.
Diet – The diet ratio, 72% microzooplankton, 28% phytoplankton is from the Chesapeake Bay model.

Microzooplankton

Q/B – A consumption ration of 350 year $^{-1}$ was estimated by Ecopath after designating a P/Q ration of 0.4, as taken from the Chesapeake Bay Model.
P/B – A total mortality rate for microzooplankton of 140 $year^{-1}$ was taken from the Chesapeake Bay model.
Biomass – Biomass was estimated based on an assumed EE of 0.95.
Diet – The 100% phytoplankton diet follows the Chesapeake Bay model.

Phytoplankton

P/B – We elected to use the Chesapeake value of 160 $year^{-1}$ over the Delaware Bay value of 60 $year^{-1}$ as the Chesapeake is a highly eutrophic system more similar to the conditions found in Barnegat Bay.

Biomass – An estimated wet weight of 7.705 t/km2 was calculated using the August 2011 to September 2012 data (ugC/L) collected as part of the Governor's Barnegat bay Initiative and a conversion ratio of 10 mg wet weight:mg C (Emax report, Dalsgaard and Pauly 1997). However, this biomass is far too small to support the grazing pressure calculated. The minimum biomass required to balance the model assuming an ecotrophic efficiency of 0.95 is 25.2 t/km^2, which is in-line with the estimates for the Chesapeake Bay.

Benthic algae

P/B – The Chesapeake model assumed a value of 80 $year^{-1}$.

Biomass – Biomass of benthic algae was estimated based on an assumed EE of 0.9 (Chesapeake).

SAV

P/B – Mortality for *Z. marina* was estimated in the Chesapeake as Z = P/B =5.11 $year^{-1}$, which was taken from a similar system in Japan (Oshima *et al.*, 1999).

Biomass – In 1979 there was approximately 8,053 ha of mapped submerged aquatic vegetation (Northern segment: 767, Central segment: 5,126, Southern segment: 2,160) out of the 27,900 hectares of Barnegat Bay (Lathrop et al 2001). The highest recorded annual eelgrass maximum biomass in the southern and central portions of the bay occurred in 2004 and was 219.7 g dry wt /m^2, while the highest Ruppia biomass recorded in the northern segment occurred in 2011 and was 32.8 g dry wt/ m^2 (Kennish et al 2013). Expanding the biomass estimates over the 1979 SAV acreage yields a baywide total biomass of 1,625.891t, or 5.82t/km^2

Appendix 2 – Ecopath Initial Diet Composition

	Piscivorous seabirds	Non-piscivorous	weakfish	striped bass	summer flounder	bluefish	winter flounder	Atlantic silversides	Atlantic croaker	spot	Atlantic menhaden
Piscivorous seabirds											
Non-piscivorous											
weakfish	0.0056			0.2		0.013					
striped bass	0.0166										
summer flounder	0.011										
bluefish	0.02										
winter flounder	0.0058				0.2						
Atlantic silversides	0.017		0.05	0.221	0.132	0.087					
Atlantic croaker			0.005	0.01		0.005					
spot			0.03			0.011					
Atlantic menhaden	0.1			0.206		0.255					
river herring	0.028										
mummichog	0.03					0.36					
bay anchovy	0.07		0.535	0.2	0.273	0.094	0.018				
benthic infauna/epifauna		0.276	0.352	0.06	0.186	0.066	0.742	0.59	0.8	0.509	0.18
amphipods			0.022				0.07	0.244		0.25	
blue crabs	0.004		0.006	0.1	0.2	0.103	0.002				
hard clams		0.01		0.003			0.157			0.057	
oysters											
copepods								0.154	0.2	0.18	0.338
Microzooplankton											
sea nettles											
ctenophores											
benthic algae											
phytoplankton											0.421
SAV		0.128									
detritus		0.011			0.009	0.006	0.011	0.012		0.004	0.061
import	0.692	0.575									

	river herring	mummichog	bay anchovy	benthic infauna	amphipods	blue crabs	hard clams	oysters	copepods	Micro zoo	sea nettles	ctenophores
Piscivorous seabirds												
Non-piscivorous												
weakfish												
striped bass												
summer flounder												
bluefish												
winter flounder												
Atlantic silversides												
Atlantic croaker												
spot												
Atlantic menhaden												
river herring												
mummichog												
bay anchovy											0.054	
benthic infauna/epifauna	0.435	0.260		0.02	0.02	0.5						
amphipods	0.055	0.170	0.044									
blue crabs						0.125						
hard clams						0.175						
oysters												
copepods	0.5	0.19	0.582								0.421	0.666
Microzooplankton			.370	0.08	0.08				0.72			0.334
sea nettles												
ctenophores											0.525	
benthic algae		0.12		0.3	0.3	0.05	0.5					
phytoplankton	0.005			0.4	0.4		0.25	0.99	0.28	1		
SAV						0.05						
detritus	0.005	0.26	0.004	0.2	0.2	0.1	0.25	0.01				
import												

Appendix 3 - Landing Calculations for the Barnegat Bay Ecopath Model

Directed Fisheries

The National Marine Fisheries Service (NMFS) commercial landings database is the most comprehensive record of commercial landings available for the time period of interest (1950-2011). However, these data represent landings for all of New Jersey, and are not Barnegat Bay specific. The NMFS landings data used below are a subset of the statewide landings based on gear that could be used within an estuary. Gear types considered usable in the bay include the following: by hand; cast nets; dip nets, common; fyke and hoop nets, fish; hand lines, other; pots and traps, blue crab; and weirs. Because these gear types have been used in the Barnegat Bay as well as other larger estuaries throughout the state (Raritan Bay, Delaware Bay, *etc.*), this subset likely overestimates commercial removals from Barnegat Bay. Where Barnegat Bay specific landings data are available they were used to the maximum extent possible.

Recreational landings for finfish were taken from the NMFS Marine Recreational Fisheries Statistics Survey (MRFSS) for Ocean County, inland waters only. The landings for 1981 were used to initialize the model as that is the earliest year for which data is available.

The source and calculations for each species are described below.

Atlantic croaker – Based on the subset of NMFS commercial landing data, there was no harvest of Atlantic croaker reported in the 1980s. There were no recreational landings of croaker reported for Ocean County.

Atlantic Menhaden - There was no commercial harvest of menhaden recorded in the NMFS landing data for the gear types used in Barnegat Bay in 1980. There were no recreational landings of menhaden reported for Ocean County in the MRFSS database. Menhaden are commonly used as bait in the recreational fishery in Barnegat Bay, therefore an estimated landing of 0.2MT was attributed to the recreational fishery, though this likely underestimates landings.

Blue Crab – In Barnegat Bay the commercial blue crab fishery can be divided into a winter dredge fishery and a pot/trap line fishery in the remainder of the year. Landings data specific to Barnegat Bay were available from the NJDEP for 1995-2011, while statewide landings were available from NMFSS for 1980-2011. The NJDEP data was regressed on the NMFS data and the results used to calculate bay specific total landings for 1981-1994. The winter dredge fishery represented approximately 17% of the baywide total (NJDEP data); this ratio was used to estimate the gear specific landings from the total baywide landings of 221 metric tons for 1981. Therefore the winter dredge fishery in 1981 landed an estimated 38.1 metric tons while the pots and trot lines accounted for an estimated 183.3 metric tons. In 2007 the recreational harvest of blue crabs in Barnegat Bay was estimated to be 80% of the total commercial harvest (B. Muffley personal communication), leading to an estimated recreational harvest of 177.1 metric tons in 1981.

Bluefish – Barnegat Bay specific commercial landings were available for bluefish for 1997 only (Kennish SCR). The bay specific landings represented 21% of the subset landings for that year (NMFS). That ratio was utilized to calculate an estimated Barnegat Bay specific commercial landing of 0.02 metric tons for 1980. In 1981approximately 209.1 metric tons of bluefish were landed in Ocean County inland waters (MRFSS).

Hard Clam – Hard clams are historically one of the most important commercial fishery resources in Barnegat Bay. Hard clam landings from Barnegat Bay approached 226.8 metric tons in 1980, the closest year for which data was available (G. Calvo, personal communication of NMFS data, 2011). There are no estimates of hard clam recreational landings available.

River herring – Alewife and blueback herring have been combined into this single category given the similarities in their life history strategies and propensity to co-migrate. In 1981 there were no commercial landings of either species in the subset landings, and no landings reported for Ocean County's recreational inland fishery. However, there were known fisheries for river herring within the bay associated with bait collection. As such a total landing of 0.1MT was assumed based on the landings in subsequent years and split evenly between the recreational and commercial sectors.

Spot –There were no commercial landings of spot recorded in the subset landing data for the late 1970s through mid 1980s. There were 1.1 metric tons of spot landed in the Ocean County inland recreational fishery in 1981.

Striped Bass – In 1981 there were no commercial landings of striped bass recorded in the subset landing data. There were no landings reported for Ocean County's recreational inland fishery. However, there was a well-documented recreational fishery present at the time, therefore 26 MT was used , which is the average of reported landings from 1981-201.

Summer flounder – Commercial landings of summer flounder approached 0.2 metric tons in 1981 according to the subset NMFS database. There were 224.4 metric tons of summer flounder landed in the Ocean County inland recreational fishery in 1981.

Weakfish - Barnegat Bay specific commercial landings were available for weakfish for 1993 only (Kennish SCR). The bay specific landings represented approximately 5.2% of the gear specific statewide landings for that year (NMFS landing data). That ratio was utilized to calculate an estimated Barnegat Bay specific commercial landing of 0.078 metric tons for 1981. There were 3.29 metric tons of weakfish landings reported for Ocean County's recreational inland fishery in 1981.

Winter flounder – The NJDEP Bureau of Marine Fisheries estimates a commercial harvest of approximately 10.68 metric tons of winter flounder from Barnegat Bay in 1981. In 1981 there were 247 metric tons of winter flounder landed in the Ocean County inland recreational fishery.

OCNGS

The Oyster Creek Nuclear Generating Station "landings" info can be divided into two categories, impingement/impingeable size losses and entrainment losses. Impingement losses describe those animals that become trapped on the traveling Ristroph screens (9mm mesh) associated with the Circulating Water Intake

Structure (CWIS) and are subsequently deposited into a fish return system and into the discharge canal. Impingeable size losses are biota that are large enough to be impinged on the Ristroph screens if they were present at the Dillution Water Intake Structure (DWIS). Entrainment losses are the biota that pass through the CWIS and DWIS structures and pass through the plant and dilution pumps, respectively. The data used to estimate these values were collected as part of periodic relicensing of the facility, and were most recently collected during 2005-2007 and include in the "Characterization of the aquatic resources and impingement and entrainment at Oyster Creek Nuclear Generating Station" September 2008.

Impingement/Impingeable size losses

During 2006-2007 the estimated annual biomass of the young of year (YOY) and older ages of selected fish and crustaceans impinged on the traveling screens at the CWIS was calculated (Appendix A: Detailed Characterization of the aquatic resources and impingement and entrainment at Oyster Creek Nuclear Generating Station, Tables A-7 and A-8). The biomass of each species was then multiplied by the empirically determined impingement mortality rate (Appendix H, Tables H-2 and H-4) to derive a CWIS impingement mortality (kg/yr). The estimated annual biomass of impingeable sized fish and shellfish that were entrained through the DWIS was calculated (Tables A-15 and A-18) and multiplied by the empirically determined mortality rates (Tables H-5 and H-6) to derive a DWIS impingeable size mortality (kg/yr). It should be pointed out that the mortality rates were instantaneous, that is injured individuals were considered "live" at the time of counting, and thus the mortality rates are likely low.

Entrainment losses

Entrainment losses occur when biota are able to avoid or slip through the traveling screens at the CWIS and are carried through the cooling water system or are taken up by the DWIS. The number of individual fish in each species entrained into either the CWIS (Table A-10) or DWIS (A-20) are broken into 5 size categories; eggs, yolk sac larvae, post-yolk sac larvae, YOY, and YOY+. Blue crabs were divided into adult, juvenile, and megalops (tables A-12 and A-22). For this model the entrainment analysis was limited to post-yolk sac larvae, YOY, and YOY+ fish and megalops stage of blue crab. Biomass for each species/size class was calculated by taking the median or mode length from the CWIS entrainment sampling length frequency histograms (Appendix C: Impingement and entrainment studies at Oyster Creek Generating Station 2005-2007) and searching the literature for the corresponding weight. This weight was multiplied by the annual estimated number of individuals to derive an estimate of annual biomass. The biomass estimate was then multiplied by the appropriate empirically determined mortality rate to derive an estimate of entrainment losses for both the CWIS and DWIS. The latent mortality was calculated as the number of live, healthy entrainable-size specimens collected from the discharges who survived for 24 hours (Appendix F, Sections 2 and 3). The mortality was applied equally across all size classes. Given that this methodology does not take into account individuals that do not survive passage through the system it likely underestimates mortality. The specific values selected for the length, weight, and mortality rate for each species are detailed below.

Adult and juvenile blue crabs were not included in the entrainment analysis as there are a number of discrepancies in the crab data. The CWIS impingement sampling collected crabs in the 8-166mm size range; these specimens should not be able to pass through the Ristroph screen, thus nearly eliminating any entrainment at the CWIS. Further, any crabs of this size should be considered part of the "entrainment of impingeable sizes" DWIS calculations, and to include them in DWIS entrainment would be double counting.

Atlantic croaker –

Post-yolk sac – Lengths ranged from 4-16mm, with a rather uniform distribution between 7-15mm. The ASMFC 2005 stock assessment for larval croaker suggests a mode of 11mm and a weight range of 0.02 – 0.04g. An average weight of 0.03g was used in the analysis.

YOY – The lengths of YOY croaker ranged from 15-72mm, with the distribution skewed heavily to the left. The modal length was 21mm. An average weight of 0.06 grams at 21mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for croaker. The empirically determined weakfish mortality rate (CWIS 0.8, DWIS 0.75) was used as they are both Sciaenids and share similar characteristics at the larval stage.

Atlantic Menhaden

Post-yolk sac – Lengths were bimodally distributed from 6 – 33 mm, with the larger mode at 24 mm. Hettler (1976) found an average weight of 0.195 grams at 28mm.

YOY – Lengths were evenly distributed between 27-42mm , with a mean length of 34. Hettler (1976) found an average weight of 0.494 grams at 34mm.

Mortality – A 24 hour mortality rate of 1 was used for the CWIS and 0.72 for the DWIS.

Atlantic silverside -

Post-yolk sac – Lengths were unimodally distributed from 4 – 8 mm, with the mode at 5mm.

YOY – Lengths were evenly distributed between 71-85mm. The silverside should be fully recruited to the Ristroph screen at 72mm, so 71mm was selected. An average weight of 0.2.25 grams at 71mm was calculated using the length-weight regression from FishBase.

YOY+ - Lengths were evenly distributed between 74-102mm, with a mean at 87mm. An average weight of 4.71 grams at 87mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for silverside. The empirically determined bay anchovy mortality rate (CWIS 0.97, DWIS 0.94) was used as they have similar body shapes and tolerances at the larval stage.

Bay anchovy -

Post-yolk sac – Lengths were unimodally distributed from 3 – 37 mm, with the mode at 8mm. Using the length-weight relationship in Table 5 of Leak and Houde (1987), an 8mm individual is approximately 11 days old, and would have a dry weight of 0.000114g. If larvae are assumed to be 95% water, this would lead to a wet weight of 0.0023

YOY – Lengths were unimodally distributed between 26-69mm , with a modal length of 34. An average weight of 0.32 grams at 34mm was calculated using the length-weight regression from FishBase.

Mortality - A 24 hour mortality rate of 0.97 was used for the CWIS and 0.94 for the DWIS.

Summer flounder –

Post-yolk sac – Lengths were unimodally distributed from 10 – 17 mm, with the mode at 14mm. An average weight of 0.04 grams at 14mm was calculated using the length-weight regression from FishBase.

YOY – Lengths were unimodally distributed between 12-17mm , with a modal length of 14. Given the overlap in lengths with post-yolk sac, it appears the demarcation between classes is based on eye migration. An average weight of 0.04 grams at 14mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for summer flounder. The empirically determined winter flounder mortality rate (CWIS 0.88, DWIS 0.90) was used as they have similar body shapes and tolerances at the larval stage.

Weakfish –

Post-yolk sac – Lengths were unimodally distributed from 2 – 14 mm, with the mode at 5mm. Using the empirically measured mean dry weight of 0.000171g for 5mm larvae from Duffy and Epifanio (1994) leads to a wet weight of 0.0034 grams assuming 95% water.

YOY – Lengths were evenly distributed between 11-123mm , with a mean length of 36. An average weight of 0.41 grams at 36mm was calculated using the length-weight regression from FishBase.

YOY+ - The only size captured in sampling was 172mm. An average weight of 0.44 grams at 172mm was calculated using the length-weight regression from FishBase.

Mortality - A 24 hour mortality rate of 0.80 was used for the CWIS and 0.75 for the DWIS.

Winter flounder –

Post-yolk sac – Lengths ranged from 2-11mm, with a relatively uniform distribution between 3-6mm. The average length was 5mm. . Based on mean larval lengths in Buckley et al. (1991), a 6mm winter flounder is approximately 4 weeks old. Laurence (1975) determined the mean dry weight of a 4 week old winter flounder kept at a similar temperature to be 0.000206g. This leads to a wet weight of 0.00412 grams assuming 95% water.

YOY – Lengths ranged between 6-7mm, with 6mm fish dominating the catch. Given the overlap in lengths with post-yolk sac, it appears the demarcation between classes is based on metamorphosis. Laurence (1975) determined the mean dry weight of a metamorphosed winter flounder to be 0.001243g. This leads to a wet weight of 0.02486 grams assuming 95% water.

Mortality - A 24 hour mortality rate of 0.88 was used for the CWIS and .90 for the DWIS.

Blue Crab –

Megalops – There was no information provided in the OCNGS reports on the length, weight, or mortality of blue crab megalopae with regard to entrainment sampling. Blue crab instar #1 have an average carapace width of 2.5mm, which is sufficiently small enough to pass through the Ristroph screen, and have an estimated average of weight of 0.0033 grams (Newcombe et al., 1949). Mortality was assumed to be similar to that found empirically for *Mysidopsis bigelowi* during the study period of 0.66 and 0.17 for the CWIS and DWIS respectively.

Fuzzy cognitive mapping in support of integrated ecosystem assessments: developing a shared conceptual model among stakeholders.

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Abstract
Ecosystem-based approaches, including integrated ecosystem assessments, are a popular methodology being used to holistically address management issues in social-ecological systems worldwide. In this study we utilized fuzzy logic cognitive mapping to develop conceptual models of a complex estuarine system among four stakeholder groups. The average number of categories in an individual map was not significantly different among groups, and there were no significant differences between the groups in the average complexity or density indices of the individual maps. When ordered by their complexity scores, eight categories contributed to the top four rankings of the stakeholder groups, with six of the categories shared by at least half of the groups. While non-metric multidimensional scaling (nMDS) analysis displayed a high degree of overlap between the individual models across groups, there was also diversity within each stakeholder group. These findings suggest that while all of the stakeholders interviewed perceive the subject ecosystem as a complex series of social and ecological interconnections, there are a core set of components that are present in most of the groups' models that are crucial in managing the system towards some desired outcome. However, the variability in the connections between these core components and the rest of the categories influences the exact nature of these outcomes. Understanding the reasons behind these differences will be critical to developing a shared conceptual model that will be acceptable to all stakeholder groups and can serve as the basis for an integrated ecosystem assessment.

Keywords: ecosystem based management, Barnegat Bay, fuzzy logic cognitive mapping, FCM,

1.0 Introduction

It is widely accepted that the sustainable management of natural resources must include consideration of human interactions with the environment, not only from a unidirectional perspective (humans impacting natural systems or vice-versa), but with the understanding that these coupled socio-ecological systems are dynamic and have a variety of two-way interactions and feedbacks (An and Lopez-Carr 2012, Liu *et al.* 2007). The realization that the use of natural resources is inextricably interwoven with the social, political, and economic complexities of human systems has led to these management challenges being called "wicked problems" (Xiang 2013), *i.e.* "problems which are ill-formulated, where the available information is confusing, where there are many clients and decision makers with conflicting values, and where the ramifications in the whole system are thoroughly confusing" (Churchman 1967). With an ever increasing number of wicked problems recognized in social-ecological systems throughout the globe (Sayer *et al.* 2013, Jentoft and Chuenpagdee 2009, Ludwig 2001) the idea of ecosystem-based management has gained traction, particularly in marine policy in the United States (NOAA 2006). Ecosystem-based management (EBM) attempts to look at a defined geographic area in a holistic manner, defining management strategies for an entire system rather than individual components (Levin *et al.* 2009).

To successfully manage resources from an ecosystem-wide perspective it is necessary to gather pertinent information on all of the system components, but by definition the data available in instances of wicked problems are confusing, as no clear patterns are readily emergent, or if there are patterns they are often contradictory. One organizing framework to synthesize and analyze large amounts of confusing data to support EBM is the Integrated Ecosystem Assessment, or IEA (Levin *et al.* 2009). The IEA approach is a series of formal processes during which relevant stakeholder groups (including public representatives, scientists, managers and policy makers) synthesize existing knowledge regarding the ecosystem in question, set ecosystem management objectives, select management options, and then adjust future management actions based on feedback from continuing monitoring. The initial activity in the IEA process is the scoping step, during which stakeholder groups define the ecosystem to be addressed, review existing information, construct a conceptual ecological model that identifies ecosystem attributes of concern and relevant stressors, and develop appropriate management objectives (Levin *et al.* 2008). Generally, this step is conducted during one or more workshops (Hobbs *et al.* 2002, McClure and Ruckelshaus 2007) where participants interact in a facilitated format designed to generate consensus on the ecosystem attributes and management objectives. However, there are concerns with the quality of both the process and the outcome when public participation is included in solving environmental issues (NRC 2008). In particular, prior studies have shown that groups tend to converge on majority views, that powerful or influential individuals or groups may attempt to dominate or unduly influence the proceedings, and that quality processes and outcomes, especially those related to consensus building, can be cost prohibitive (NRC 2008).

In light of the potential problems described above, there is a clear need for a strategy that can combine traditional scientific knowledge with public local context, thereby reducing uncertainty and providing for a diversified and adaptable knowledge base (Raymond *et al.* 2010, Gray *et al.* 2012). One methodology that has been suggested is Fuzzy Logic Cognitive Maps (FCMs) (Axelrod 1976). FCM are a simplified way of mathematically modeling a complex system (Özesmi and Özesmi 2004), and have been used to represent both individual and group knowledge (Gray *et al.* 2012). This approach has been applied to processes and decisions in human social systems, the operation of electronic networks, and in the ecological realm to identify the interactions between social systems, biotic, and abiotic factors in lakes (Özesmi 2003, Hobbs *et al.* 2002), coal mine environs (Zhang *et al.* 2013), farming systems (Vanwindekens *et al.* 2013), nearshore coastal zones (Meliadou *et al.* 2012, Kontogianni *et al.* 2012) and the summer flounder fishery (Gray *et al.* 2012), but applications in estuaries has been rare.

In this paper we investigate if fuzzy logic cognitive mapping can be used to develop a shared conceptual model among various stakeholder groups that can serve as the basis for an integrated ecosystem assessment in a complex estuarine system. We first develop conceptual ecosystem models for different stakeholder groups using FCM. Next we combine those models into a shared conceptual ecosystem model. A shared understanding of the important components and processes of the ecosystem in question is critical if stakeholder groups are to fully “buy-in” to future management decisions (Ogden *et al.* 2005). The FCM methodology ameliorates many of the challenges associated with integrating the different types of stakeholder knowledge (Gray *et al.* 2012), and the transparent nature of the model combination allows stakeholders to identify how each groups’ model contributes to the overall understanding. We do not expect the different groups’ conceptual models to share all of the components; rather we anticipate these differences to be highly informative. Indeed, understanding why these differences occur is likely to help us avoid misunderstandings and disagreements during future phases of the IEA process (Kontogianni *et al.* 2012b). Therefore, we analyze the components and structural similarities and differences among the models to assess the utility of this approach as the basis for the IEA scoping process, with the understanding that the scoping process is an essential first step toward effective EBM.

2. Methodology

2.1 Study Site

The social ecological system we have chosen to study is the Barnegat Bay, a 279 km^2 lagoonal estuary located in central New Jersey, USA (Figure 1). The surrounding 1,730 km^2 watershed is home to an estimated 580,000 year round residents (US Census Bureau 2012), with a summer population that swells to over 1 million with the influx of tourists. The physical setting of the watershed is well described by Kennish (2001), but points germane to our study are repeated here. Land use is a mix of urban and suburban uses in the northeast and along the barrier islands, grading to less sparsely populated forested areas to the south and west. Portions of the E.B. Forsythe National Wildlife Refuge and the Pinelands National Reserve are located along the eastern and western sides of the watershed, respectively. There is limited extractive and agricultural land use, and other than minor hard clam and blue crab fisheries, no real commercial fishing. The watershed is considered "highly eutrophic" (Bricker *et al.* 2007), mainly due to nutrient enrichment through non-point source pollution, and the nation's oldest continuously operating nuclear power plant, Oyster Creek Nuclear Generating Station, is located within the watershed. There is extensive recreational use of the bay's waters for fishing, boating, sailing, and to a lesser degree, bathing.

2.2 Data collection

FCMs are models of a how a system operates based on key components and their causal relationships. The components can be tangible aspects of the environment (a biotic feature such as fish or an abiotic factor such as salinity) or an abstract concept such as aesthetic value. The individual participants identify the components of the system that are important to them, and then link them with weighted, directional arrows. The weighting can range from -1 to +1 (Hobbs *et al.* 2002, Özesmi and Özesmi 2004, Gray *et al.* 2012), and represents the amount of influence (positive or negative), that one component has on another.

To collect FCM from a wide variety of stakeholders with knowledge of the Barnegat Bay ecosystem we contacted the Barnegat Bay Partnership, a US Environmental Protection Agency National Estuary Program, to obtain a list of their management and science committee members, as well as a list of public citizens who have expressed long-term interest in the ecosystem. While the map of an individual stakeholder provides information regarding that particular individual's conception of the important components and linkages within the system, it can be combined with other individuals within the group to produce a more robust picture of the group's understanding of the system (Özesmi and Özesmi 2004). In addition, all of the individual stakeholder maps can be combined into a single map depicting the collective understanding of the system. To this end, the individuals were divided into four groups that were determined *a priori*: scientists (n=19), managers (n=11), environmental non-governmental organizations (n=6), and local residents (n=6) (Table 1). These groups were selected to represent several (though not all) of the major categories of stakeholders present in ongoing efforts to manage and improve the bay's natural resources. The scientist group consisted of individuals from academia, state, and federal institutions who have conducted research within the Barnegat Bay watershed, while managers were from federal, state, county, or local natural resource management agencies who had jurisdiction on some form of activity within the watershed. Environmental non-governmental organizations included local, statewide, and regional groups who are active in watershed protection. The local residents were referred to us by other interviewees, and included commercial and recreational fisherman as well as private citizens with a long-standing interest in the bay.

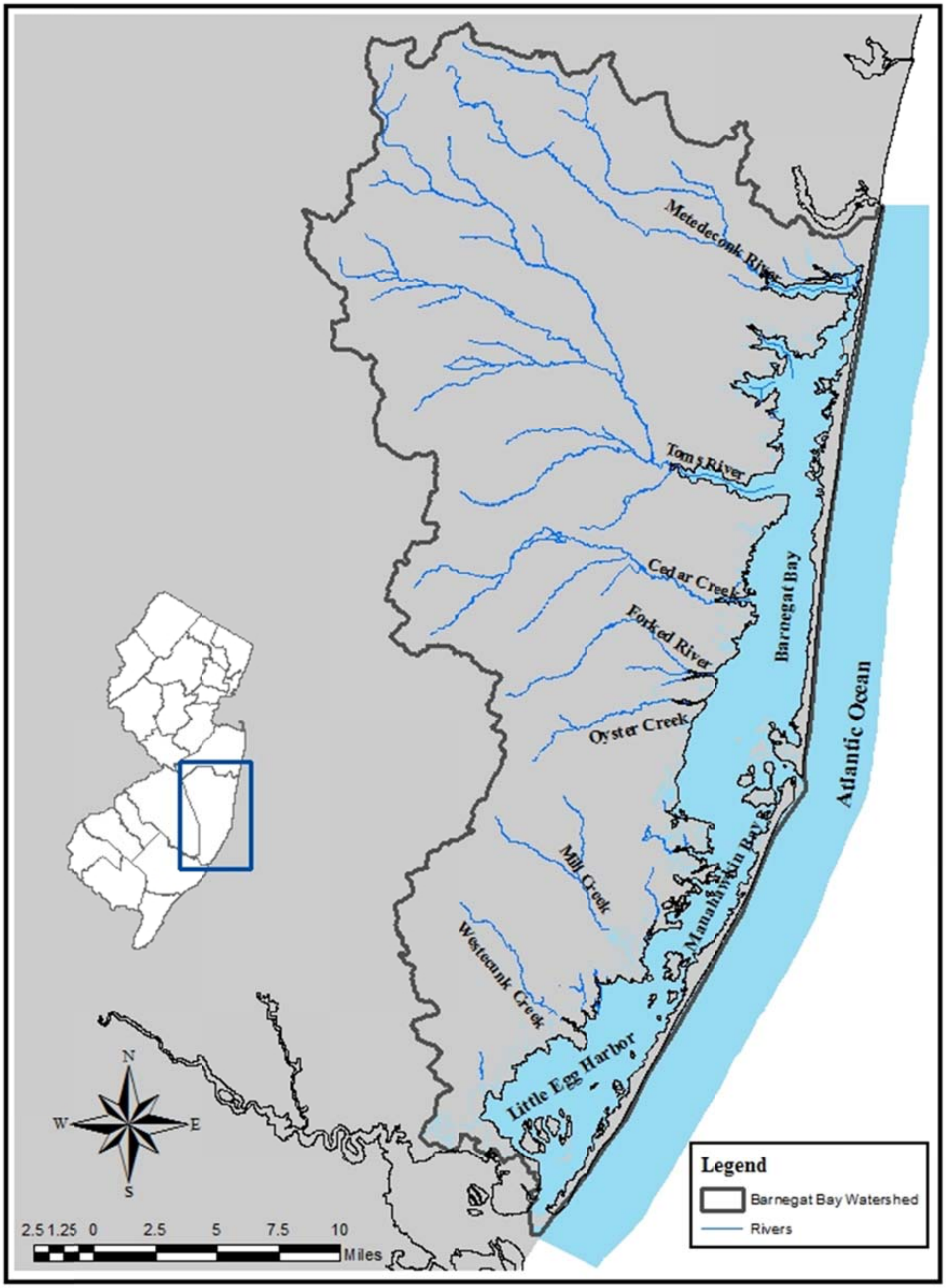
Table 1: Information on stakeholders who completed fuzzy cognitive maps on the Barnegat Bay social-ecological system			
Stakeholder group	Maps (N)	People (N)	Occupation/organization/social group
Scientists	19	19	Academic scientists, federal and state agency research scientist
Managers	11	11	Federal, state, county, and local resource managers
Environmental NGOs	6	6	Regional, statewide, and local environmental non-profits
Local people	6	6	Baymen, commercial fisherman, recreational fisherman, longtime (+40 year) residents

In accordance with the procedures used in prior studies (Carley and Palmquist 1992, Özesmi and Özesmi 2004, Gray *et al.* 2012) individuals were interviewed separately, and each interview began with an overview of the project, a promise of anonymity, and an example of a simple FCM related to an issue outside of the realm of ecology, namely traffic flow. Interviewees were then asked to describe what they considered to be the key components of the Barnegat Bay social-ecological system and how those components relate to one another. They were then asked to score the strength and direction of the relationship using positive or negative; high, medium, or low. The discussion continued until the interviewee was satisfied that the map as drawn accurately depicted their understanding of the system. This ranged anywhere from 45 minutes to 180 minutes, with the typical session lasting 90 minutes. Once mapping was complete, the interviewees were asked which of the components in their maps they would like to see increased and which decreased. The interviews were conducted under an approved human subjects protocol (number: E13-560).

2.3 Data Analysis

There are a number of different methods that can be used to analyze the data contained within an FCM, many of which are based upon graph theory (Harary *et al.* 1965, Özesmi and Özesmi 2004, Kosko 1991). To better understand the structure of an individual FCM we translated each map into a square adjacency matrix, with all of the variables acting as potential transmitters (influencing other variables) v_i on the vertical axis and the same set of variables acting as receivers (influenced by other variables) v_j on the horizontal axis (see Supplemental Figure 1 for an example). A list of all individual variables mentioned throughout the process was compiled and redundant variables (plurals, different names for the same species, *etc.*) were eliminated. When two variables represented opposite directions of the same concept (*i.e.* dam construction and dam removal) the more prevalent variable was retained and the other variable was renamed, with the polarity of the interactions reversed, in keeping with accepted practices (Kim and Lee, 1998). The interactions strengths between variables were then scored, with high interactions scored as 0.75, medium as 0.5, and low as 0.25 (Harary *et al.* 1965).

Figure 1 – Map of Barnegat Bay watershed with New Jersey inset.



To more easily understand the components and patterns within an individual FCM it is often helpful to simplify the map by reducing the number of variables (Harary *et al.* 1965). After all of the maps were completed we listed the full set of variables and identified those most often mentioned. We then subjectively combined less frequently mentioned variables into larger categories based on shared characteristics, a process known as qualitative aggregation. For example, "homes", "urban development", "housing", and "overdevelopment", were combined, with a number of other similar variables, into a category called "development".

With the large list of variables reduced into broader categories, the type of categories, and number of each, were identified to provide additional insight into the overall structure of the map and how these categories relate to each other (Bougon *et al.* 1977, Eden *et al.* 1992, Harary *et al.* 1965). Each category was classified as transmitter, receiver, or ordinary (both influenced by and influencing other categories), based on its indegree and/or outdegree (Table 2). Indegree is the cumulative strength of the connections entering the category (sum of the absolute values within a column in the matrix), while outdegree is the cumulative strength of the connections exiting the category (sum of the absolute values within a row in the matrix) (Özesmi and Özesmi 2004). A transmitter category has positive outdegree and no indegree, a receiver category has no outdegree and a positive indegree, and an ordinary category has positive indegrees and outdegrees (Bougon *et al.* 1977). Finally, the centrality, or a measure of a category's connectedness to other categories within the map, as well as the overall strength of those connections, was calculated as the sum of the indegree and outdegree values of a given category (Harary *et al.* 1965).

Table 2: Fuzzy Cognitive Map Indices

Term	Definition
Indegree	Cumulative strength (absolute value) of the connections entering a category
Outdegree	Cumulative strength (absolute value) of the connections exiting a category
Centrality	Sum of the indegree and outdegree for a given category
Receiver	A category with a positive indegree and no outdegree
Transmitter	A category with a on indegree and a positive outdegree
Ordinary	A category with positive indegree and outdegree
Complexity	The ratio of receiver categories to transmitter categories within a map (R/T)
Density	The number of connections within a map divided by the total connections possible between categories (C/N^2)

Indices of complexity and density were also determined for each stakeholder map. The complexity of a map is calculated as the ratio of receiver categories to transmitter categories (R/T). A large number of receiver categories in a map suggests a system where there are multiple outcomes (Eden *et al.* 1992), while a large number of transmitter categories suggest that a system is hierarchical in nature, and driven by "top down" thinking (Özesmi and Özesmi 2004). Density describes how well connected categories are within the map, and is determined by dividing the number of connections present by the maximum number of connections possible (Hage and Harary, 1983). A dense map suggests that an interviewee (or stakeholder group) perceives a number of possible pathways to influence a variable in their map (Özesmi and Özesmi 2004).

In addition to developing indices for each individual map, maps were combined 1) within stakeholder groups to produce four group maps and 2) across all individuals to produce a community map. To combine maps the connection values between two given categories are added, so connections represented in multiple maps are reinforced (provided they have similar signs) while less common connections are not reinforced, but are still included in the map (Özesmi and Özesmi 2004). To compare connection values across group maps, the summed values are divided by the number of individuals in the group.

Non-metric multidimensional scaling (nMDS) was used to assess the similarities between individual stakeholder maps (R v3.0.2). This technique orders samples by rank similarity along their two most important latent gradients and has an advantage over other ordination techniques in that it has a greater ability to accurately represent complex relations among samples in two-dimensional space (Clarke and Warwick 2001). The nMDS data were calculated as each category's centrality score for an individual stakeholder and then the Bray Curtis index was used to construct the sample similarity matrix (variable by stakeholder array). The nMDS plot was then visually assessed to identify patterns between stakeholder groupings.

Besides understanding the structure of the stakeholder groups' and community maps, maintaining the initial conditions through time allows us to determine if the model will coalesce around a stable state, go into a limit cycle, or enter into a chaotic pattern (Dickerson and Kosko 1994). To generate this steady state, the adjacency matrix of the cognitive map is multiplied by an initial steady state vector (a value of 1 for each element of the vector). The resulting vector is then subject to transformation using a logistic expression ($1/(1 + e^{-1\times x})$) to bound the results in the interval [0,1] (Kosko 1987). This new vector is then multiplied by the original adjacency matrix and again subject to the logistic function, repeating these steps until an end result is reached.

If the model reaches a steady state outcome, it is then possible to run hypothetical "what-if" scenarios to compare the function of the various models. The hypothetical scenario developed for our simulation was to maintain the category "development" at 0, which is a possible policy prescription, albeit a potentially unpopular one. To do this we utilize the process described above to determine the stable state, but this time the value of the category "development" in the vector is maintained at 0 in each time step. Setting the value of a category of interest in the multiplication vector between 0 and 1 at each time step was referred to as "clamping" by Kosko (1986). The difference between the values of the final vector of the clamped procedure compared to the steady state vector describe the relative change to the conceptual system given the framework provided by each stakeholder group. A conceptual schematics of map aggregation and steady state calculations are provided in Supplemental Figure 1 and a flow diagram of the steps in the data analysis process is provided as Supplemental Figure 2.

3.0 Results

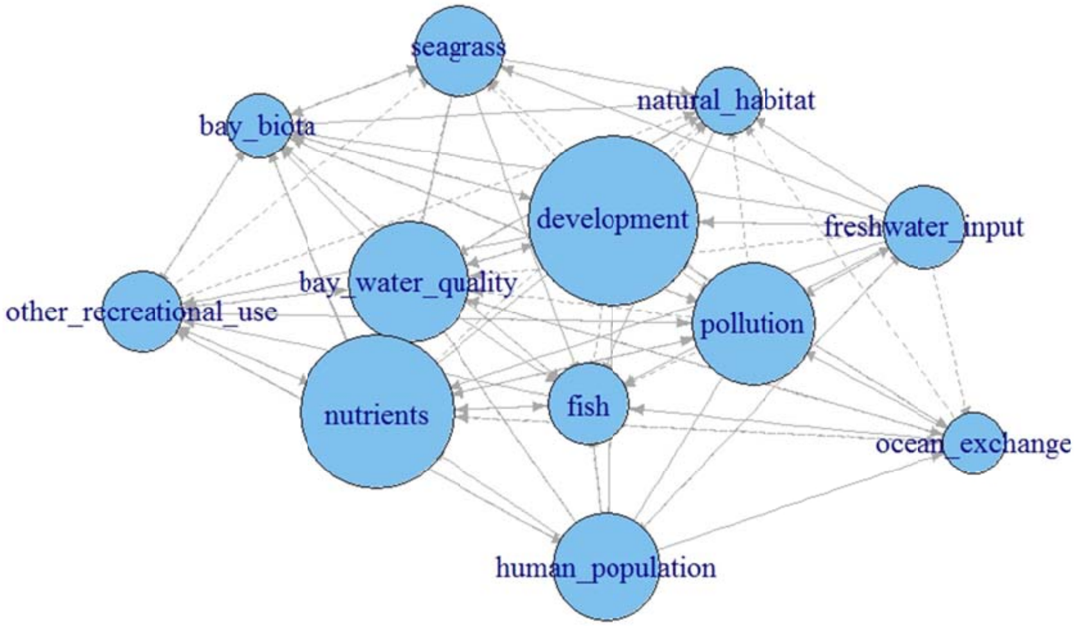
We created fuzzy cognitive maps for 42 individuals from the four targeted stakeholder groups (Table 1). The stakeholders identified 346 unique variables as important to understanding the Barnegat Bay social – ecological system, which were then aggregated into 84 categories for further analysis. Individual maps contained an average of 25 variables, which when aggregated led to an average of approximately 20 categories per map. The average number of categories in an individual map was not significantly different among groups, with the exception of NGOs ($p = 0.02$), who had an average of nearly 30 categories per map (Table 3). An examination of the accumulation curves for the total number of categories versus the number of interviews shows that the managers and scientists were well sampled, while the NGO and local residents' curves had not yet flattened out (Supplemental Figure 3). All of the NGOs active in the watershed at the time of the study were interviewed, limiting the number of samples of available. The pool of potential interviewees who met the criteria for the local resident group was also limited in size. However, the trajectories of these two groups is similar to that of the scientists and managers, suggesting that few new categories would have been added through additional interviews.

There were no significant differences between the groups in the average complexity (df=38, p=0.492) or density (df= 38, p=.129) indices of the individual maps (Table 3). The environmental NGOs and local residents had slightly higher complexity scores (more receiver categories) than the other two groups, while the managers and scientists had slightly higher average densities. The community map, by definition, contained the full suite of categories, but had an order of magnitude more connections than the group maps, leading to a map with the most interconnections between categories, and therefore the highest density. The increased number of interconnections in the community map led to all of the categories being classified as "ordinary" (i.e., both a transmitter and a receiver), with the exception of biodiversity, which was a receiver category. A subset of the community map that includes the categories with centrality scores greater than one, and their interconnections, is shown in Figure 2. For a complete list of all variables and their centrality scores please see Table S1 in the supplemental information.

Table 3: Graph indices by stakeholder group. All values, except for number of maps, are mean and standard deviation.

	Scientists	Managers	Environmental NGOs	Local people	Community
Maps	19	11	6	6	42
Number of categories (N)	20.6 (4.3)	21.2 (5.3)	29.8 (13.4)	19.3 (3.6)	84
Number of transmitter categories (*T*)	5.1 (2.7)	4.4 (2.7)	5.8 (3.3)	4.7 (2.5)	0
Number of receiver categories (*R*)	3.2 (2.8)	2.3 (1.9)	4.5 (2.9)	4.3 (1.8)	1
Number of ordinary categories	12.3 (4.3)	14.5 (4.0)	19.5 (10.8)	10.3 (2.7)	83
Number of connections (C)	38.3 (13.3)	49 (17.8)	64 (40.7)	29.5 (9.3)	1071
C/N	1.9 (0.5)	2.3 (0.6)	2.1 (0.5)	1.5 (0.4)	12.75
Complexity (*R/T*)	0.7 (0.8)	0.6 (0.5)	0.9 (0.5)	1.1 (0.6)	
Density	0.09 (0.03)	0.11 (0.04)	0.08 (0.03)	0.08 (0.02)	0.15

Figure 2: Subset of the community conceptual model. The twelve nodes with centrality scores greater than 1.0 are shown. Node size is related to centrality score, solid lines are positive interaction strengths, dotted lines are negative interactions strengths.



When ordered according to their centrality scores, eight different categories contributed to the top 4 rankings of the stakeholder groups, and six of the categories were shared by at least half of the groups (Table 4). Development had the strongest interactions for managers and local residents and was second only to nutrients for scientists and NGOs. Pollution, bay water quality, seagrass, and human population were also key shared categories, though the strength of the interactions, and their ranking, varied between groups. The outdegree strength for development and human population was at least two times that of the indegree, while pollution and bay water quality had indegrees slightly larger than outdegrees. The direction and magnitude of the strengths for seagrass varied between groups, with local residents giving it a moderately larger outdegree and scientists scoring the indegree twice as high.

Table 4: Category centrality scores by stakeholder group. Centrality is the sum of the indegree and outdegree for each category and is an index of its connectedness to other variables within the map. The categories included below represent the top four categories of each stakeholder group.

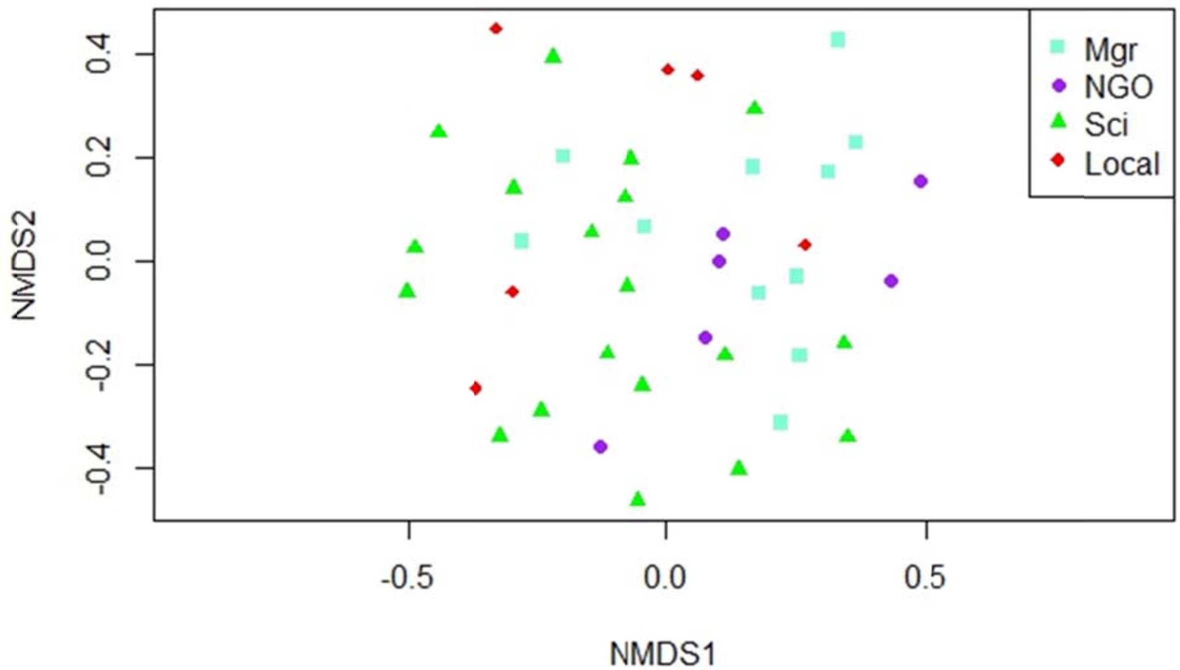
	Scientists	Managers	Environmental NGOs	Local people	Community
Development	1.91	3.93	3.50	3.0	2.75
Human population		3.15		2.48	
Bay ecological condition				2.25	
Seagrass	1.68			1.92	

Bay water quality		3.27	2.75		1.96
Nutrients	3.10		4.25		2.48
Pollution		3.03	3.29		2.00
Fish	1.33				

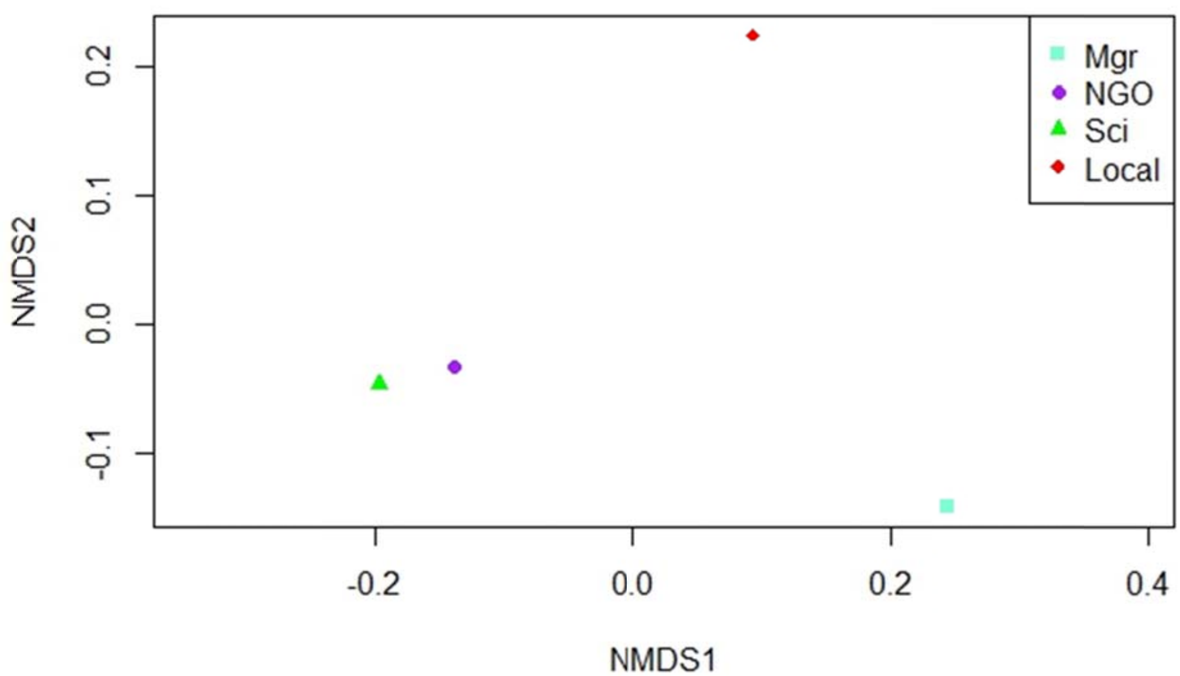
There was substantial overlap in nMDS space between the individual cognitive maps of scientists and all other groups, moderate overlap among managers and NGOs and local residents, and little overlap between NGOs and local residents (Figure 3a). The individuals within each stakeholder group were spread along both nMDS axes, indicating that there is a diversity of conceptual models within each group. When viewed as aggregated stakeholder groups, the Scientist and NGO conceptual models are most similar, while the others are quite dissimilar (Figure 3b).

Figure 3: nMDS plot of the a) individual and b) stakeholder group fuzzy cognitive maps based on centrality scores. Because nMDS is a non-metric procedure, the axes labeled NMDS1 and NMDS2 have no units associated with them. Stress values were 0.279 and 0.169, respectively. Stakeholder groups include Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and Local residents (Local).

a)

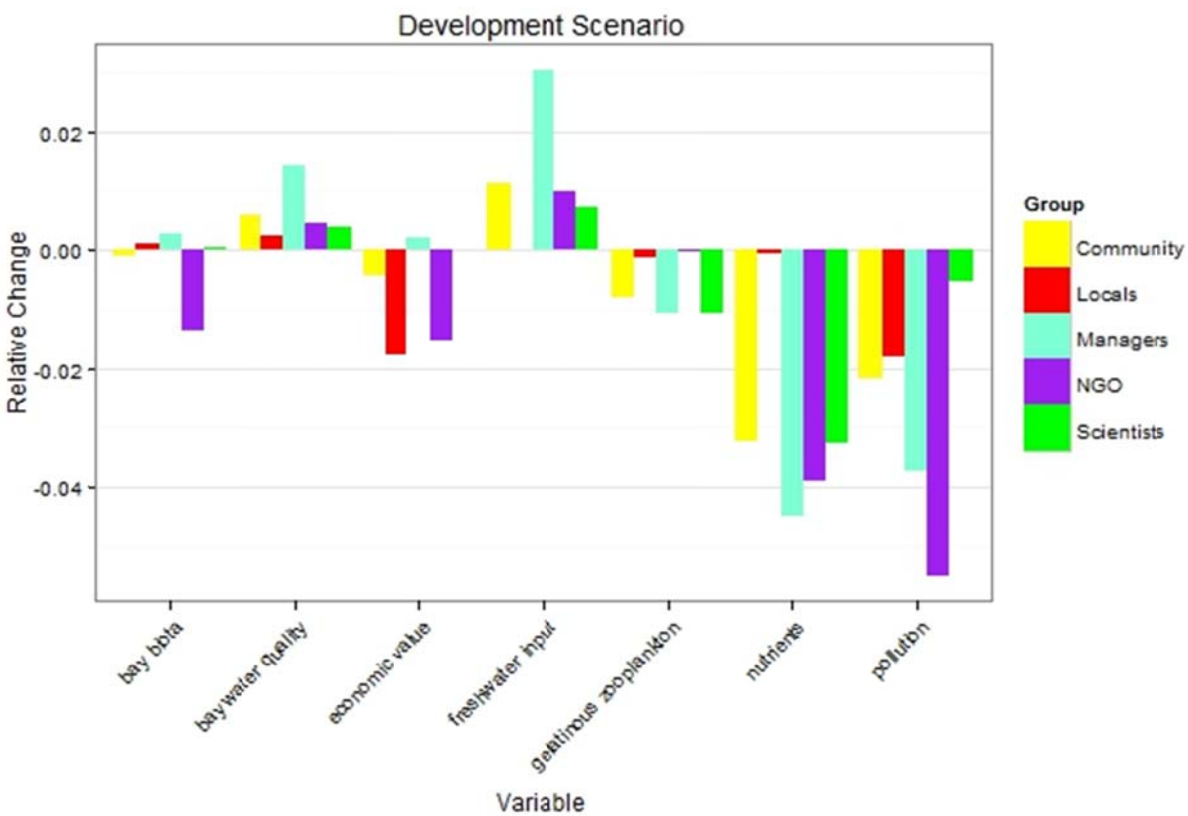


b)



The hypothetical scenario model run further elucidated similarities and differences between the conceptual models of the stakeholder groups (Fig 4). When development was clamped to a low level, nutrients and pollution, two of the more central categories in all groups' models, both decreased compared to the steady state models, though the degree of decline varied among groups. The declines in these two categories were driven primarily by the direct linkages participants made between them and development. The increase in bay water quality and decrease in gelatinous zooplankton (primarily identified by participants as the nuisance jellyfish *Chrysaora quinquecirrha*, or stinging sea nettle) across all groups' models appears to be driven by a number of indirect linkages to development. In the case of bay water quality, one potential pathway identified was a decrease in development leading to a decrease in impervious surfaces, which lead to a decrease in runoff, which improved bay water quality. While the prior examples showed concurrence in the effects of low development across the groups' models, they differed in the outcome of the economic value category; the NGOs' and locals' models predicted a decrease in economic value associated with a decrease in development, while the managers' models predicted an increase in economic value.

Figure 4: Results of the scenario model when development was clamped to a low level. Relative change along the y-axis is the difference between the "low development" scenario compared to the initial steady-state solution for a given category. Stakeholder group models were constructed for Local residents (Local), Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and an aggregate of all cognitive maps (Community).



4.0 Discussion

4.1 The applicability of FCMs in estuarine environments

Fuzzy cognitive maps have been used to model stakeholder perceptions of causal relationships in social-ecological systems in a variety of settings (Özesmi and Özesmi 2003, Meliadou *et al.* 2012, Gray *et al.* 2012, Kontogianni *et al.* 2012, Vanwindekens *et al.* 2013, Zhang *et al.* 2013). This study is the first to apply the methodology to an estuarine ecosystem. Estuaries are both an ecosystem in their own right as well as an ecotone between terrestrial and aquatic and between freshwater and the ocean. Thus, we might expect that people's perceptions of estuaries could be more heterogeneous than FCMs of other systems. The complexity of estuaries is reflected in the large number of unique variables mentioned by the stakeholders during the creation of their FCMs. While caution should be used when comparing FCM indices between studies due to potential differences in methodology (Eden *et al.* 1992), the number of variables recorded in this study exceeds those compiled using similar methods for a large lacustrine system (Özesmi and Özesmi 2003) and a nearshore coastal region (Meliadou *et al.* 2012). This level of detail was not driven by a small number of stakeholders in any particular group; the mean number of categories per map, complexity, and density were all similar across groups, suggesting that all of the stakeholders recognize the complexity and multidimensionality of estuaries.

A potential downside to this is the resulting intricacy of the overall community model, which still includes 84 categories after aggregation. Jørgensen (1994) theorized that quantitative ecological models have a bell-shaped curve in regard to performance verses complexity, and others have suggested that cognitive maps are most easily interpreted when the number of variables ranges from the low teens (Buede and Ferrell 1993) to 30 (Özesmi and Özesmi 2004). Due to its semi-quantitative nature it is difficult to determine how close a FCM approximates the realities of the social–ecological system. However, the models developed here reach a stable state during the scenario analysis in less than 10 iterations and generally follow well established ecological theory, providing additional support for the validity of the findings.

While fuzzy cognitive mapping is robust enough to handle the large number of variables associated with a complex ecosystem, the applicability of this technique is constrained by how well (or poorly) it handles non-monotonic responses (Carvalho 2013). This is particularly true for temperate estuaries, where long gradients in environmental factors like temperature and salinity can lead to dome-shaped response curves. Many of the interviewees attempted to side-step this issue by framing the response in terms of what they anticipated the departure from the current range of the condition would be. For example, interviewees said that increased temperature would lead to an increase in the abundance of a given biota (through some physiological or habitat mediated mechanism) up to some degree, after which increasing temperatures would lead to decreases in abundance. They then posited that it would be unlikely that temperatures in the estuary would ever exceed the inflection point, and thus the overall response is positive. This solution is similar to that previously identified by Hobbs *et al.* (2002) in their construction of an FCM for Lake Erie. Differences in an individual's interpretation on how best to address non-monotonic responses likely led to conflicting causal relationships when aggregating FCMs for the community map. Thus the response of some categories to changes in the scenario model is dampened, though based on notes taken during the interview process it would be limited to a few biotic components and the strength of the interactions tended to be low.

4.2 Differences in stakeholder cognitive models

To develop a comprehensive management plan for complex systems a shared understanding of the components among the stakeholders is a prerequisite (Ogden *et al.* 2005). The findings of this study suggest that while all of the stakeholders interviewed perceive the Barnegat Bay ecosystem as a complex series of social and ecological interconnections and shared common structural elements, there are differences in the components and linkages of their aggregated conceptual models which influence the final state of the system. There is a core set of components that are present in most of the stakeholder groups' FCMs and have high centrality scores; the stakeholder groups all agree that these components are crucial in managing the system towards some desired outcome. However, the number and strength of linkages between these key components and the rest of the social-ecological system varies, such that the FCMs of two stakeholder groups can have opposite outcomes. This was seen in the scenario modeling, where low levels of development through time led to an increase in the economic value of the bay in the Manager's FCM and a decrease in economic value in the NGO and Local models.

One potential reason for the opposing results in the group models may be the primary focus of the groups themselves, including their conception of the relevant "social" dimensions of the system. The individuals comprising the Manager group are tasked with regulating the use of the biological resources of the estuary (fish, crabs, clams, birds), and in their maps a decrease in development yields an increase in biomass and a concomitant increase in economic value through commercial harvest or other recreational opportunities. In contrast, the environmental NGOs often take a broadly anthropocentric view of the social-ecological interactions of the estuary, and their maps contained social and political actors that were not mentioned by others. These social concepts (taxes, land price) often had strongly positive relationships between development and economic value.

While the aggregated community map incorporates multiple perspectives, and thus should be a more complete representation of the system (Gray *et al.* 2012), being able to articulate where, and why, stakeholder groups may have similar or diverging views on important causal relationships will be critical to developing the consensus approach needed to plan appropriate management actions for protection and restoration. A starting point for understanding the convergences or divergences is seen in the arrangement of the group maps in the nMDS, which suggests that the scientists and NGOs place similar importance on a broad variety of categories. This stands in contrast with the managers and local residents, who do not share similar centrality scores among categories. Thus one would expect, and should plan for, the additional effort that will be required to bring these two groups to consensus.

4.3 Further FCM benefits

Opposite interactions (positive versus negative) between two components shared across groups' conceptual models may reflect differences of opinion or perspective but also may point to areas where the understanding of the relationships between concepts is incomplete, such as the effects of climate change on biodiversity and species invasions, and changes to the bay's water quality associated with changes in freshwater input. The identification of these knowledge gaps through FMCs combined with the management objectives developed during the initial stages of the integrated ecosystem assessment will allow for a prioritization of future research and funding needs. These divergences may also indicate subjects where more recent scientific findings have not yet been widely incorporated by those outside specific fields of study (*i.e.* saltmarsh – nutrient interactions, biochemical and physical induced changes in nutrient loads, the pathway and flow of nutrients around the bay) and therefore where additional education/outreach may be warranted. Additionally, the community map can assist in the selection of variables for monitoring once a course of actions has been agreed upon. Given a modeled scenario, or suite of scenarios, the components along the causal chain can be identified, eliminating potential indicators that are not responsive to the management efforts proposed, or do not meet the criteria for informative indicators (Rice and Rochet, 2005). This is particularly important in an age of shrinking research budgets and results-focused management at resource agencies.

5.0 Conclusion

We have shown that Fuzzy Cognitive Mapping can be a useful tool for organizing the intricate connections between social and ecological concepts within a highly complex ecosystem, and when applied across stakeholder groups can elucidate not only those mechanisms for which there is a shared understanding, but also highlight where additional resources should be focused to gain the greatest insights into system operation. While subject to limitations associated with representing non-monotonic response variables, they can nevertheless serve as a basis from which the initial steps of an Integrated Ecosystem Assessment can proceed. In particular, the individual interview procedure utilized herein avoids some of the pitfalls associated with group participation in the scoping process and provides a clear scaffolding upon which potential management and policy scenarios can be evaluated.

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7.0 References

An, L. and Lopez-Carr, D. 2012. Understanding human decisions in coupled natural and human systems. Ecological Modelling 229:1-4.

Axelrod, R., 1976. Structure of Decision: The Cognitive Maps of Political Elites. Princeton University Press, Princeton, NJ, USA.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. Effects of Nutrient Enrichment In the Nation's Estuaries: A Decade of Change. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD. 328 pp.

Bougon, M., Weick, K., Binkhorst, D., 1977. Cognition in organizations: an analysis of the Utrecht Jazz Orchestra. Administrative Science Quarterly. 22, 606–639.

Buede, D.M., Ferrell, D.O., 1993. Convergence in problem solving: a prelude to quantitative analysis. IEEE Transactions On Systems Man and Cybernetics 23, 746–765.

Carley, K., Palmquist, M., 1992. Extracting, representing, and analyzing mental models. Social Forces 70, 601–636.

Carvalho, J.P. 2013. On the semantics and the use of fuzzy cognitive maps and dynamic cognitive maps in social sciences. Fuzzy Sets and Systems. 214:6-19.

Churchman, C.W. 1967. Wicked Problems. Management Science. 14(4) B141-142.

Clarke, K.R., and R.M. Warwick. 2001. Change in marine communities: An approach to statistical analysis and interpretation, 2nd ed. Plymouth: PRIMER-E.
Dickerson, J.A. and Kosko, B. 1994 Virtual Worlds as Fuzzy Cognitive Maps. Presence. 3(2): 173-189.

Eden, C., Ackerman, F., Cropper, S., 1992. The analysis of cause maps. Journal of Management Studies 29, 309–323.

Gray, S., Chan, A., Clark, D., Jordan, R. 2012. Modeling the integration of stakeholder knowledge in social–ecological decision-making: Benefits and limitations to knowledge diversity. Ecological Modeling. 229:88-96

Hage, P., Harary, F., 1983. Structural Models in Anthropology. Oxford University Press, New York.

Harary, F., Norman, R.Z., Cartwright, D., 1965. Structural Models: An Introduction to the Theory of Directed Graphs. John Wiley & Sons, New York.

Hobbs, B.F., Ludsin, S.A., Knight, R.L., Ryan, P.A., Biberhofer, J., Ciborowski, J.J.H., 2002. Fuzzy Cognitive Mapping as a Tool to Define Management Objectives for Complex Ecosystems. Ecological Applications 12, 1548-1565.

Jentoft, S. and Chuenpagdee, R. 2009. Fisheries and coastal governance as a wicked problem. Marine Policy 33:553–560

Jørgensen, S.E., 1994. Fundamentals of Ecological Modelling. Elsevier, New York, 628 pp.

Kennish, M.J. 2001. Physical description of the Barnegat Bay – Little Egg Harbor estuarine system. Journal of Coastal Research Special Issue 32: 13-27

Kontogianni, A., Papageorgiou, E., Salomatina, L., Skourtos, M., and Zanou, B. 2012. Risks for the Black Sea marine environment as perceived by Ukrainian stakeholders: A fuzzy cognitive mapping application. Ocean & Coastal Management. 62:34-42.

Kontogianni, A., Papageorgiou, E., and Tourkolias, C. 2012b. How do you perceive environmental change? Fuzzy Cognitive Mapping informing stakeholder analysis for environmental policy making and non-market valuation. Applied Soft Computing, 12:3725-3735.

Kosko, B., 1986. Fuzzy cognitive maps. International Journal of Man–Machine Studies. 1, 65–75.

Kosko, B., 1987. Adaptive inference in fuzzy knowledge networks. In: Proceedings of the First IEEE International Conference on Neural Networks (ICNN-86), San Diego, CA, pp. 261–268.

Kosko, B., 1991. Neural Networks and Fuzzy Systems. Prentice-Hall, Englewood Cliffs, NJ, USA.

Levin, P.S., M.J. Fogarty, G.C. Matlock, and M. Ernst. 2008. Integrated ecosystem assessments. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-NWFSC-92, 20 p.

Levin, P.S., Fogarty, M.J., Murawski, S.A., Fluharty, D., 2009. Integrated Ecosystem Assessments: Developing the Scientific Basis for Ecosystem-Based Management of the Ocean. PLoS Biology 7, 0023-0028.

Liu, J., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C.L., Schneider, S.H., Taylor, W.W., 2007. Complexity of coupled human and natural systems. Science 317, 1513–1516.

Ludwig, D. 2001. The Era of Management Is Over. Ecosystems 4: 758–764.

McClure M, Ruckelshaus M. 2007. Collaborative science: Moving ecosystem-based management forward in Puget Sound. Fisheries 32: 458.

Meliadou, A., Santoro, F., Nader, M.R., Dagher, M.A., Indary, S.A., Salloum, B.A. 2012. Prioritising coastal zone management issues through fuzzy cognitive mapping approach. Journal of Environmental Management. 97:56-68.

National Oceanic and Atmospheric Administration. 2006. Evolving an ecosystem approach to science and management through NOAA and its partners. Available: http://www.sab.noaa.gov/Reports/eETT_Final_1006.pdf. Accessed 6 January 2014.

National Research Council. 2008. *Public Participation in Environmental Assessment and Decision Making*. Panel on Public Participation in Environmental Assessment and Decision Making, Thomas Dietz and Paul C. Stern, eds. Committee on the Human Dimensions of Global Change. Division of Behavioral and Social Sciences and Education. Washington, DC: The National Academies Press.

Ogden, J.C., Davis, S.M., Jacobs, K.J., Barnes, T., Fling, H.E., 2005. The Use of Conceptual Ecological Models to Guide Ecosystem Restoration in South Florida. Wetlands 25, 795-809.

Özesmi, U., Özesmi, S.L., 2003. A Participatory Approach to Ecosystem Conservation: Fuzzy Cognitive Maps and Stakeholder Group Analysis in Uluabat Lake, Turkey. Environmental Management 31, 518-531.

Özesmi, U., Özesmi, S.L., 2004. Ecological models based on people's knowledge: a multi-step fuzzy cognitive mapping approach. Ecological Modelling 176, 43-64.

Raymond, C.M., Fazey, J., Reed, M.S., Stringer, L.C., Robinson, G.M., Evely, A.C., 2010. Integrating local and scientific knowledge for environmental management. Journal of Environmental Management 91, 1766-1777.

Rice, J.C., Rochet, M.-J., 2005. A framework for selecting a suite of indicators for fisheries management. ICES Journal of Marine Science, 62, 516-527.

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, JL., Sheilb,D., Meijaard, E., Venter, M., Boedhihartono, A.K., Day, M., Garcia, C., van Oosten C., and Buck, L.E. 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proceedings of the National Academy of Science 110(21): 8349–8356.

United States Census Bureau. 2012. State and County QuickFacts. http://quickfacts.census.gov/qfd/states/34/34029.html. Accessed 09, January 2014.

Vanwindekens, F.M., Stilmant, D., Baret, P.V. 2013. Development of a broadened cognitive mapping approach for analysing systems of practices in social–ecological systems. Ecological Modelling. 250:352– 362

Xiang, W. 2013. Working with wicked problems in socio-ecological systems: Awareness, acceptance, and adaptation. Landscape and Urban Planning. 110(1-4).

Zhang, H., Song, J., Su, C., He, M. 2013. Human attitudes in environmental management: Fuzzy Cognitive Maps and policy option simulations analysis for a coal-mine ecosystem in China. Journal of Environmental Management. 115:227-234.

8.0 Supplemental Information

Table S1: Centrality scores by stakeholder group cognitive models. A blank value indicates a category not included in that particular group's model. The Community model is the aggregate of all individual models.

Category	Scientist	Manager	NGO	Local residents	Community
agriculture	0.34	0.20	0.08		0.22
algal blooms	0.25	0.18	0.54	0.25	0.27
atmospheric deposition	0.43	0.64	1.12		0.43
bay biota	0.61	1.32	2.35	0.71	1.04
bay ecological condition	0.30	1.02	0.50	2.25	0.71
bay salinity	0.99	0.57	1.48	0.38	0.82
bay water quality	1.04	3.27	2.75	1.88	1.96
bay water temperature	0.78	0.80	1.92	0.42	0.71
benthic biota	0.96			0.25	0.47
benthic infauna	0.41				0.19
biochemical/physical processes	0.86		0.17	0.13	0.41
biodiversity	0.12	0.20	0.25		0.11
birds	0.20	0.09	0.54	0.79	0.30
blue crabs	0.33	0.34	0.50	0.54	0.39
boating	0.91	0.70	1.04	1.27	0.88
bulkheading/docks	0.57	0.86	0.71	0.71	0.61
climate change	0.59	1.07	1.37		0.71
commercial fishing	0.28	1.10	0.13	0.13	0.44
conservation	0.03	0.77	0.13	0.88	0.29
depth	0.24	0.07	0.50	0.25	0.16
development	1.91	3.93	3.50	3.00	2.75
dissolved oxygen	0.80	0.33	0.79	0.75	0.63
dredging	0.20		0.25	0.25	0.16
economic value	0.37	1.49	0.88	0.50	0.75
ecosystem services		0.68	0.21		0.21
effective management	0.24	0.78	2.16		0.62
elected officials			1.24	0.50	0.25
erosion	0.28	0.18	0.54	0.25	0.27
fish	1.33	1.39	1.54	1.50	1.33
fishing	0.58	1.02	1.75	0.38	0.81
freshwater input	1.13	2.44	2.15	0.13	1.34
freshwater quality	0.33	0.72	1.33	0.75	0.61
freshwater use	0.50	1.07	1.42	0.38	0.71
gelatinous zooplankton	1.05	0.39	1.33	0.63	0.86

Table S1: Centrality scores by stakeholder group cognitive models. A blank value indicates a category not included in that particular group's model. The Community model is the aggregate of all individual models.

Category	Scientist	Manager	NGO	Local residents	Community
geomorphological processes	0.29	0.47	0.17		0.27
government	0.04	0.60	1.46	0.38	0.34
hard clams	0.38	0.66	0.38	0.50	0.47
harmful algal blooms	0.45	0.32	0.25		0.31
household inputs	0.30	0.39	0.25	1.00	0.42
human population	0.88	3.15	1.50	2.48	1.74
impervious surfaces	0.22	1.09	1.96		0.67
intangible values	0.17	0.86	0.42	0.38	0.38
invasive species	0.18	0.51		0.29	0.25
larval supply	0.50	0.32	0.17		0.33
macroalgae	0.18	0.11	0.46	0.88	0.30
microbial loop	0.41		0.33		0.23
natural habitat	0.99	1.64	1.27	0.38	1.08
NGOs			1.19	0.54	0.25
nutrients	3.10	2.10	4.25	0.63	2.48
ocean exchange	1.31	1.18	1.63	0.25	1.00
OCNGS	0.49	0.66	1.83	0.08	0.60
other crustaceans		0.18		1.13	0.21
other groups		0.36	0.34		0.12
other land use	0.58	0.84	1.33	0.38	0.68
other plankton	0.22		0.54	0.25	0.21
other recreational use	1.25	1.62	1.00	1.88	1.32
oysters	0.16		0.29	0.38	0.17
phytoplankton	1.27	0.40			0.64
policy decisions	0.13	1.50	0.46	0.13	0.47
pollution	1.32	3.03	3.29	1.63	2.00
precipitation	0.16	0.12	0.46		0.17
preserved open space	0.33	1.30	1.04	0.50	0.71
public	0.17	0.41	1.04		0.33
public awareness	0.20	0.91	1.08	1.58	0.68
recreational fishing	0.28	0.68		0.13	0.27
regulations	0.30	0.30	0.63	0.25	0.32
residence time	0.59	0.98	0.58		0.58
resource users	0.04		1.92		0.29
runoff	0.53	0.39	1.17	0.63	0.60
salt marshes	0.59	0.59	0.17	0.38	0.48
scientists			1.33		0.19
seagrass	1.68	1.00	1.17	1.92	1.46
sediment	0.73	0.34			0.42

Table S1: Centrality scores by stakeholder group cognitive models. A blank value indicates a category not included in that particular group's model. The Community model is the aggregate of all individual models.					
Category	Scientist	Manager	NGO	Local residents	Community
sewer systems	0.07	0.08	1.08	0.63	0.26
shellfish	0.70	0.66	0.88	0.71	0.67
stormwater	0.12	0.57	0.13	0.13	0.24
tides	0.33	0.32	0.54		0.27
tourism	0.09	1.23	1.88	0.25	0.67
turbidity	0.93	0.18	0.54	0.50	0.62
vehicles	0.07	0.50	0.67	0.42	0.32
water circulation	0.74	0.30	0.25	0.25	0.41
wetlands	0.04	1.03	0.21		0.32
wind	0.13		0.29	0.13	0.12
zooplankton	0.64	0.16	0.38	0.25	0.42

Figure S1. Conceptual schematic of the FCM combination process and steady state calculation.

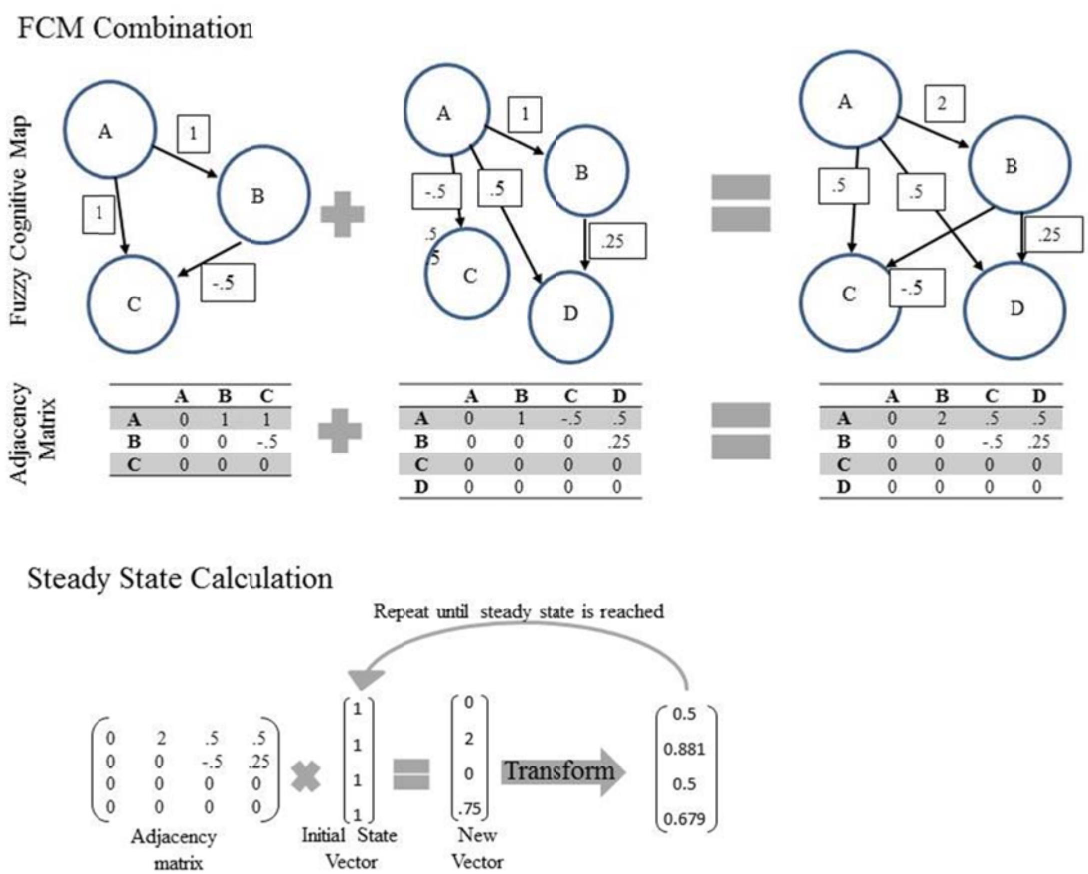


Figure S2. A flow diagram of the data analysis steps.

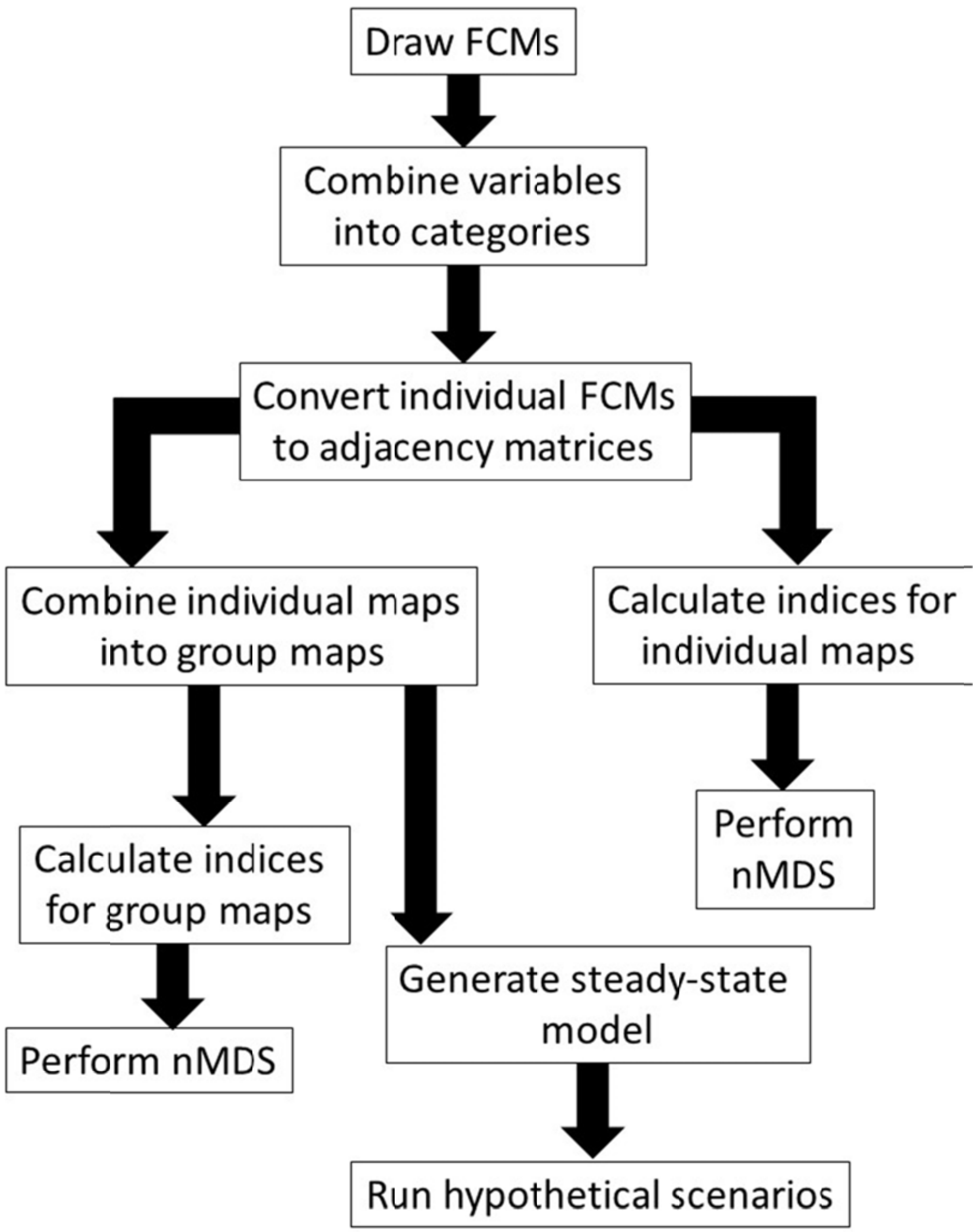
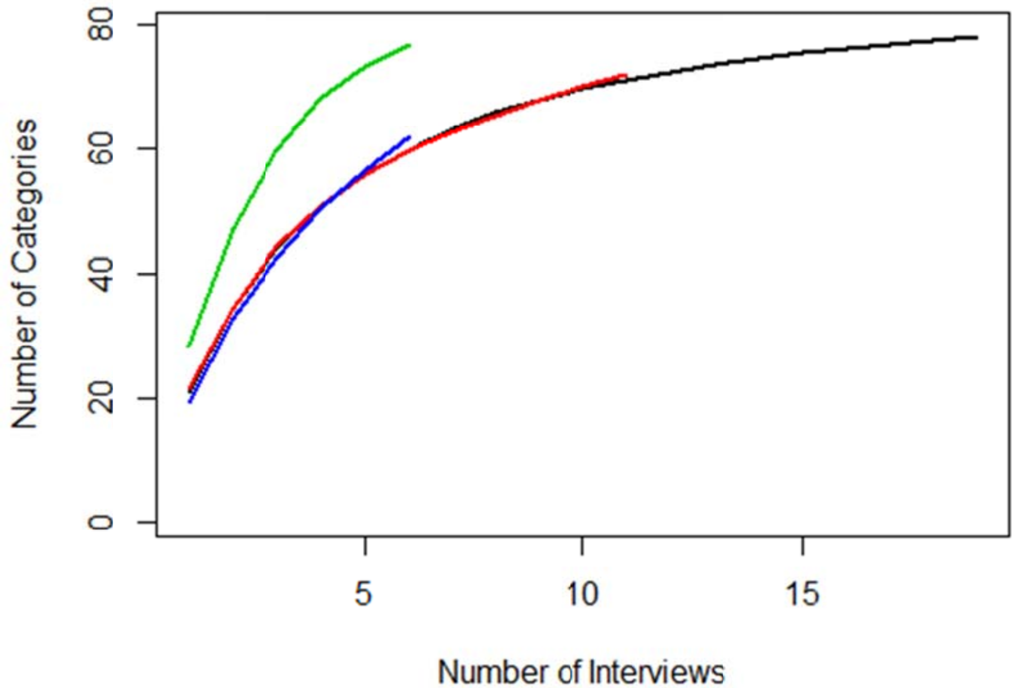


Figure S3. Accumulation curves for the total number of categories versus the number of interviews. The black line is scientists, red is managers, blue is local people, and green in environmental NGOs.



Black and white figures for printing

Figure 1 – Map of Barnegat Bay watershed with New Jersey inset.

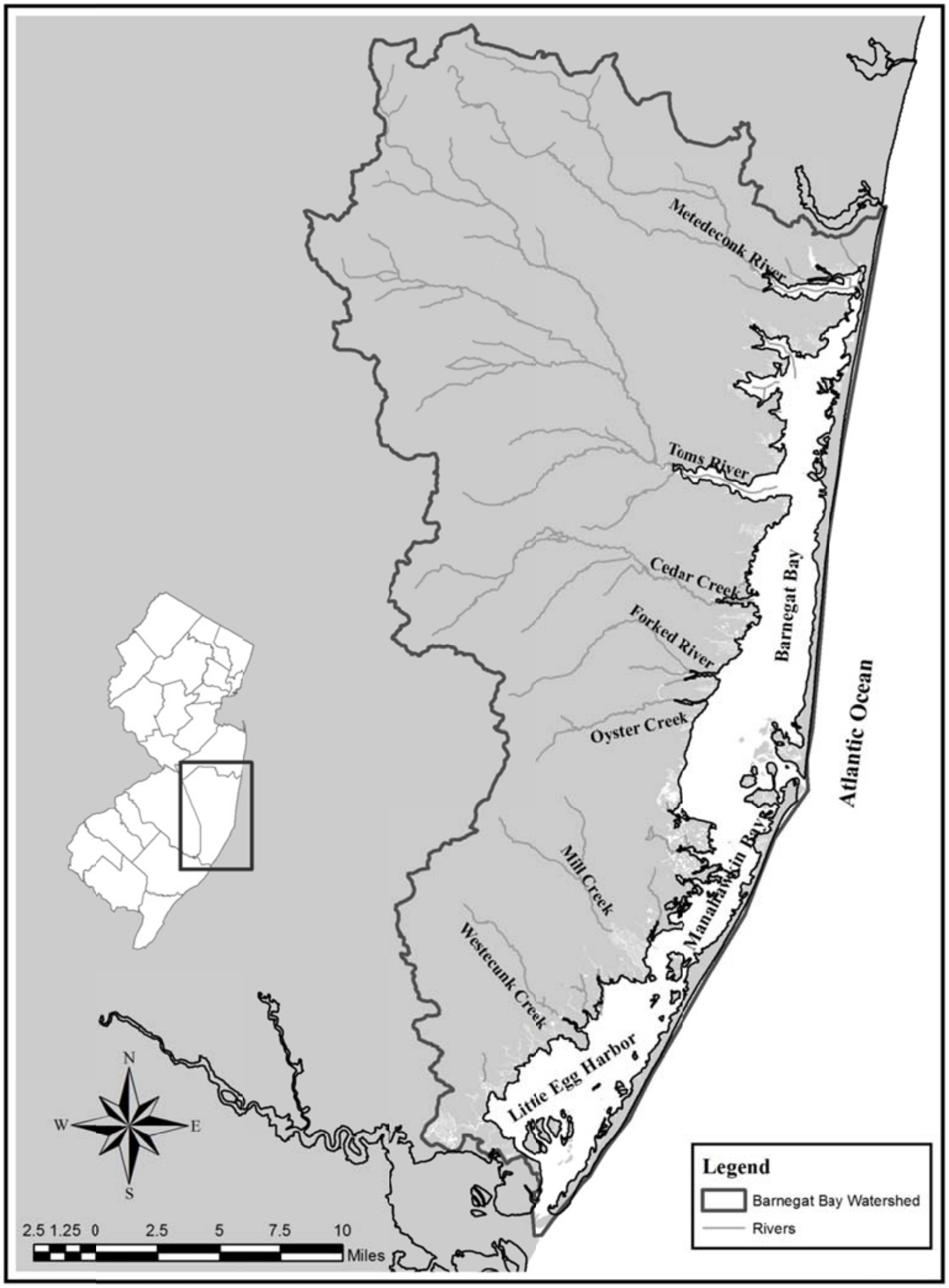


Figure 2: Subset of the community conceptual model. The twelve nodes with centrality scores greater than 1.0 are shown. Node size is related to centrality score, solid lines are positive interaction strengths, dotted lines are negative interactions strengths.

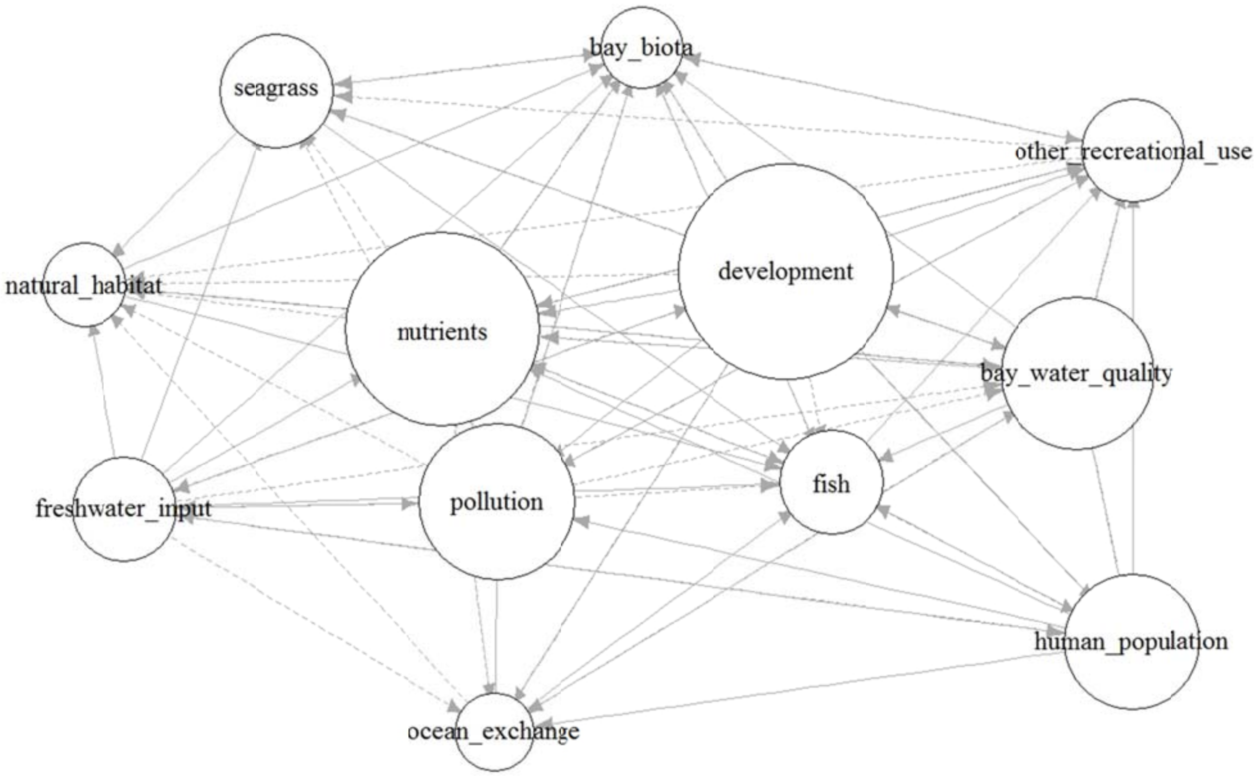
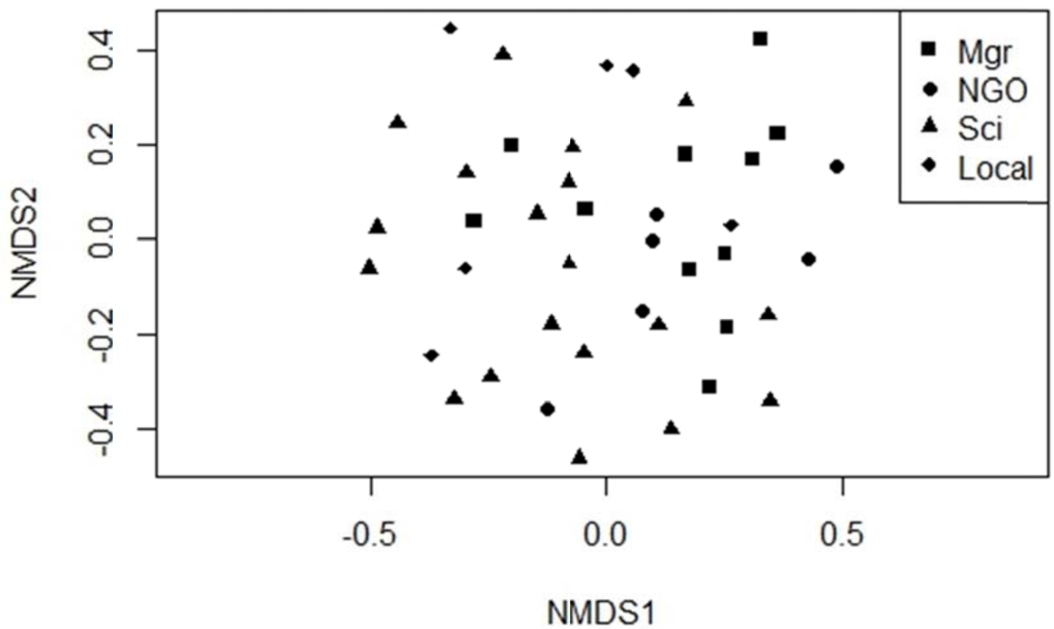


Figure 3: nMDS plot of the a) individual and b) stakeholder group fuzzy cognitive maps based on centrality scores. Stress values were 0.279 and 0.169, respectively. Stakeholder groups include Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and Local residents (Local).

a)



b)

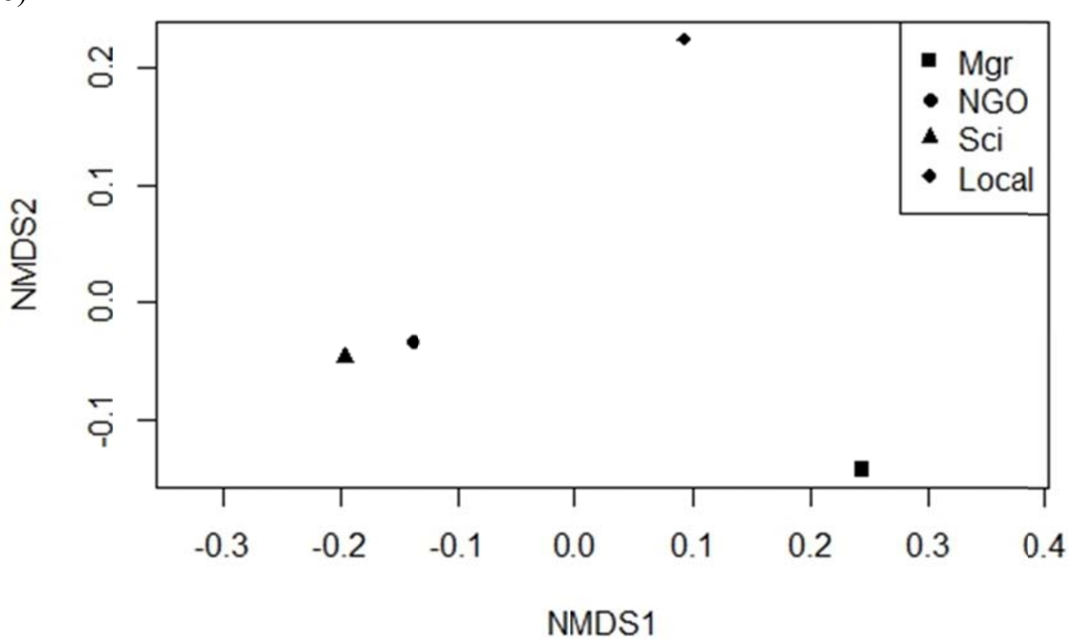
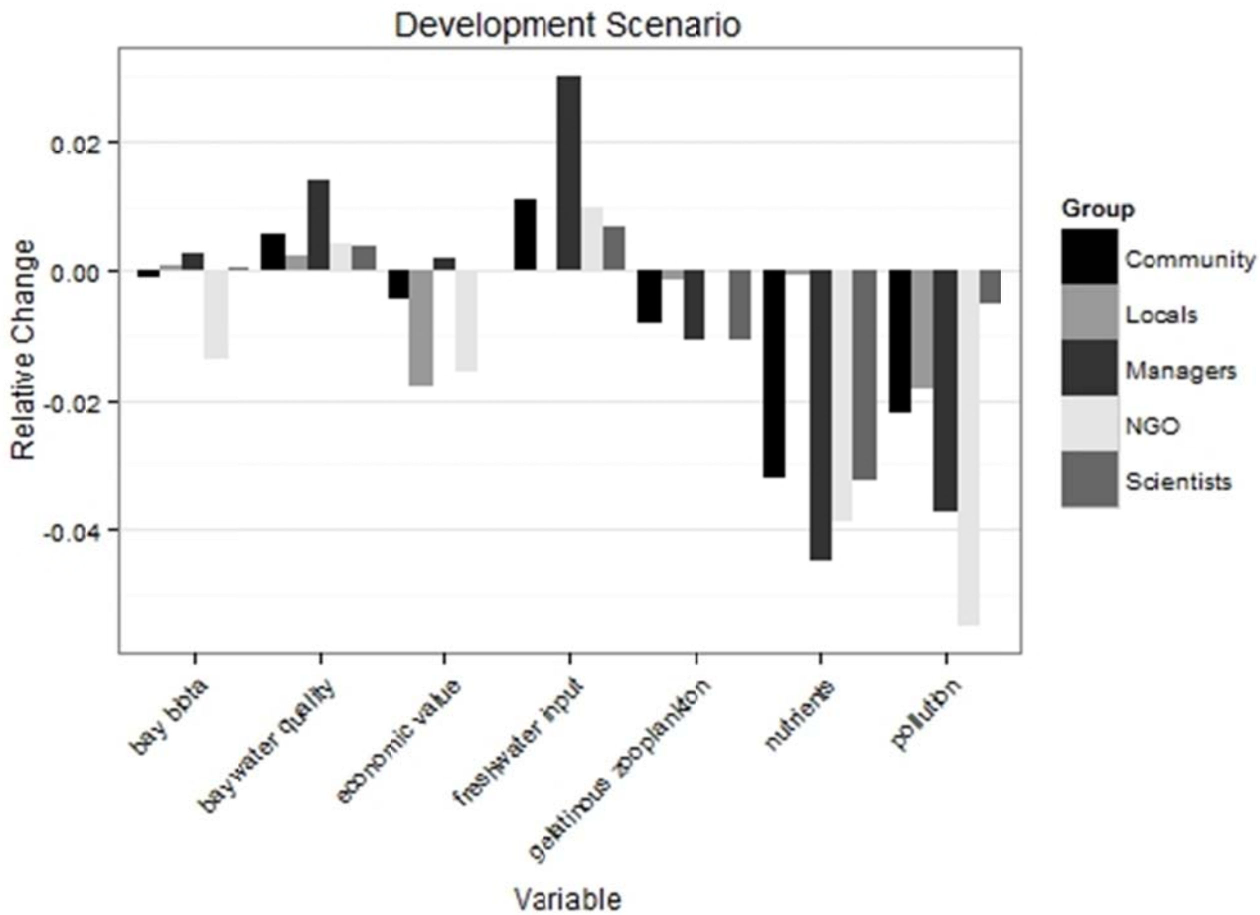


Figure 4: Results of the scenario model when development was clamped to a low level. Relative change along the y-axis is the difference between the "low development" scenario compared to the initial steady-state solution for a given category. Stakeholder group models were constructed for Local residents (Local), Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and an aggregate of all cognitive maps (Community).



Year 2 Project Report for “Multi-Trophic Level Modeling of Barnegat Bay”

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Objective 1: Refine conceptual model developed through interviews with Barnegat Bay scientists and managers

The conceptual model developed through stakeholder interviews during Year 1 of the project was further refined and analyzed during the first half of the current grant period and a manuscript was submitted to the Department for their review. Approval for submission was obtained from the Department on March 11, 2014, and the manuscript was subsequently sent to the Journal of Environmental Management. The manuscript received brief but supportive comments from a single reviewer. We revised the manuscript accordingly and it is currently under review at the journal. The submitted version is included here as Appendix 4.

Objective 2 - Incorporate Year 1 data and model parameters from NJDEP funded projects for use in the NPZ and EwE models

EwE model parameters

Ecopath with Ecosim (EwE) is a software modeling tool used to quantitatively evaluate trophic interactions within an ecosystem in order to assess options for ecosystem-based management of fisheries. The first step in the process is to develop a mass-balance model (Ecopath), which requires four groups of basic input parameters to be entered into the model for each of the species (or groups) of interest: diet composition, biomass accumulation, net migration, and catch (for fished species). Three of the following four additional input parameters must also be input: biomass, production/biomass (Z), consumption/biomass, and ecotrophic efficiency. The model uses the input data along with algorithms and a routine for matrix inversion to estimate any missing basic parameters so that mass balance is achieved.

For the purposes of the Barnegat Bay model we have set biomass accumulation and net migration to zero for all of our species groups. This is equivalent to the assumption that biomass of all species groups was at equilibrium. This is a typical assumption in the absence of information to the contrary. The biomass, production/biomass, consumption/biomass, and Ecotrophic efficiency values for the model can be found in Table 1 below. These parameters were estimated from a variety of sources, the details of which can be found in Appendix 1. The diet composition matrix can be found in Appendix 2, with the source data also listed in Appendix 1. Harvest data for recreationally and commercially important species can also be incorporated into the EcoPath model as the landings (t/km^2/year) for the year in which the model is initiated. The landings values included in the model can be found in Table 2 below, with commentary on their derivations found in Appendix 3.

As identified in Appendix 1, many of the parameters utilized in the model at this time were not developed specifically for Barnegat Bay. This is particularly true for the biomass estimates, where with the exceptions of SAV, hard clams, bay anchovy, sea nettles, and ctenophores, the other values were primarily estimated by the software, or modified from Chesapeake Bay values (seabirds). The SAV, hard clam, and bay anchovy biomass values were from studies conducted around the time of the initial year of the model. The ctenophore and sea nettle biomasses were estimated using data from the first year of the NJDEP Barnegat Bay field research projects. We attempted to utilize data from the first year of the NJDEP Barnegat Bay field research projects in combination with other Barnegat Bay specific studies for phytoplankton and amphipods but the biomass estimated by these studies was substantially less than that required to support the remainder of the model. We will revisit these estimates as additional years of Barnegat Bay specific data become available. We are also in the process of completing a simple stock assessment model to estimate biomass for blue crab given their importance to the recreational and commercial fishery sector. If successful this value will be utilized in place of the software derived estimate.

Table 1: Basic parameters for the Barnegat bay Ecosystem Model. Values estimated by Ecopath are shown in *italics*. Estimated from a variety of sources as described in Appendix 1.

Group name	Biomass (t/km^2)	Prod./biomass ($year^{-1}$)	Cons./biomass ($year^{-1}$)	Ecotrophic Efficiency	Prod./Cons.
Piscivorous seabirds	0.250	0.163	120	*0.0*	*0.001*
Non-piscivorous seabirds	0.121	0.511	120	*0.0*	*0.004*
Weakfish	*4.472*	0.260	3	0.9	*0.087*
Striped bass	*1.642*	0.4	2.4	0.9	*0.167*
Summer flounder	*2.3*	0.52	2.6	0.95	*0.200*
Bluefish	*2.733*	0.52	3.1	0.95	*0.168*
Winter flounder	*4.661*	0.52	3.4	0.95	*0.153*
Atlantic silversides	*4.741*	0.8	*4*	0.95	0.2
Atlantic croaker	*0.196*	0.916	4.2	0.9	*0.218*
Spot	*0.617*	0.9	6.2	0.9	*0.145*
Atlantic menhaden	*12.697*	0.5	31.42	0.95	*0.016*
River herring	*1.180*	0.75	8.4	0.95	*0.089*
Mummichog	*3.465*	1.2	3.65	0.95	*0.329*
Bay anchovy	4.860	3	9.7	*0.98*	*0.309*
Benthic infauna/epifauna	*81.025*	2	*10*	0.9	0.2
Amphipods	*3.438*	3.8	*19*	0.9	0.2
Blue crabs	*6.366*	1.21	4	0.95	*0.303*
Hard clams	26.18	1.681	5.1	*0.185*	*0.330*
Oyster	0.001	0.630	2	*0*	*0.315*
Copepods	*15.505*	25	*83.333*	0.95	0.3
Microzooplankton	*8.343*	140	*350*	0.95	0.4
Sea nettles	1.380	13	20	*0.077*	*0.650*
Ctenophores	7.860	16.2	35	*0.114*	*0.463*
Benthic algae	*4.614*	80		*0.900*	
Phytoplantkon	*25.221*	160		0.95	
SAV	5.820	5.11		*0.105*	
Detritus	1			*0.110*	

Table 2: Landings values used in the 1981 Barnegat Bay Ecopath model. All values are in tons/km^2/yr. Sources and calculations can be found in Appendix 3.

Group name	crab - recreational	crab pot and trap	crab winter dredge	commercial clam	OCNGS	jellyfishers	weakfish	striped bass
Piscivorous seabirds								
Non-piscivorous seabirds								
Weakfish					0.026182		0.01208	
Striped bass								0.0931
Summer flounder					0.001699			
Bluefish					2.15E-05			
Winter flounder					0.007052			
Atlantic silversides					0.024835			
Atlantic Croaker					0.013108			
Spot								
Atlantic Menhaden					0.057949			
River herring					0.000742			
Mummichog					7.00E-07			
Bay anchovy					0.011175			
Benthic infauna/epifauna								
Amphipods								
Blue crabs	0.634767	0.656989	0.136559		0.011571			
Hard clams				0.8129				
Oyster								
Copepods								
Microzooplankton								
Sea nettles						1.38		
Ctenophores								
Benthic algae								
Phytoplantkon								
SAV								
Detritus								
Sum	0.634767	0.656989	0.136559	0.8129	0.154333	1.38	0.01208	0.0931

Table 2 cont'd: Landings values used in the 1981 Barnegat Bay Ecopath model. All values are in tons/km^2/yr. Sources and calculations can be found in Appendix 3.								
Group name	summer flounder	bluefish	winter flounder	croaker	spot	menhaden	river herring	Total
Piscivorous seabirds								
Non-piscivorous seabirds								
Weakfish								0.038262
Striped bass								0.0931
Summer flounder	0.804717							0.806416
Bluefish		0.750072						0.750094
Winter flounder			0.9253					0.932352
Atlantic silversides								0.024835
Atlantic Croaker				0.0001				0.013108
Spot					0.00398			0.00398
Atlantic Menhaden						0.000716		0.058665
River herring							0.000358	0.0011
Mummichog								7.00E-07
Bay anchovy								0.011175
Benthic infauna/epifauna								0
Amphipods								0
Blue crabs								1.439886
Hard clams								0.8129
Oyster								0
Copepods								0
Microzooplankton								0
Sea nettles								1.38
Ctenophores								0
Benthic algae								0
Phytoplantkon								0
SAV								0
Detritus								0
Sum	0.804717	0.750072	0.9253	0	0.00398	0.000716	0.000358	6.365871

EwE time series data

Once the Ecopath model has been balanced the mass-balanced linear equations are then re-expressed as coupled differential equations so that they can be used by the Ecosim module to simulate what happens to the species groups over time (Christensen and Walters, 2004). Model runs are compared with time-series data and the closest fit is chosen to represent the system. Time-series data for model calibration are thus essential for developing and validating an Ecosim model (Christensen *et al*. 2009). Therefore, time-series data depicting trends in relative and absolute biomass, fishing effort by gear type, fishing and total mortality rates, and catches for as long a period as possible should be viewed as additional data requirements.

In addition to the commercial and recreational landings information as described in Appendix 2 there are few other time-series data available specific to Barnegat Bay. Many other ecosystem models glean data from formal stock assessments, which utilize similar time series data for single species management plans. Unfortunately there are no stock assessments specific to the Barnegat Bay. We have utilized the commercial blue crab landings data gathered by the NJDEP to create gear specific time series which were converted to effort and used to force the model. We are in the process of completing a simple stock assessment model to estimate a time series of biomasses for blue crab specific to Barnegat Bay, and will include that as a separate time series if successful.

We have acquired a long-term (1988-2011, except 1991-1995) otter trawl data set from the Rutgers Marine Field Station that includes 6 regularly sampled sites located in Little Egg Harbor. The CPUEs generated from this data are useful for fitting to overall trends. We have performed a trawl efficiency study for the DEP sponsored survey, which utilizes the same gear as this survey. Trawl efficiency estimates account for the fact that not all individuals within the path of the trawl are captured. Efficiency estimates will allow us to develop baywide biomass estimates from the current survey data, which we can then use to fit the time series endpoints. The results of the trawl efficiency study are presented here under Objective 3 revised.

In addition to the fish and crab data referenced above, the NJDEP has hard clam surveys from 1986/1987 in Barnegat Bay and Little Egg Harbor, 2001 in Little Egg Harbor, 2011 in Little Egg Harbor, and 2012 in Barnegat Bay. The 1986/1987, 2001, and 2011 data are incorporated into the model. Release of the

2012 data was delayed due to the effects of Hurricane Sandy. This data will be incorporated when it becomes available.

SAV coverage for the bay is available for 1980, 1987, 1999, 2003, and 2009 based on aerial photograph analysis in Lathrop et al. (2001) and Lathrop and Haag (2011). The acreage of seagrass in each year serves as a datapoint of relative abundance. Limited data was available for benthic algae and a time series was not able to be developed.

The last source of Barnegat Bay specific time series data comes from OCNGS. Because of the nature of OCNGS operations, the cooling and dilution intake structures function as an on/off type activity, with the only shutdowns associated with temporary, short term maintenance. As such the plant flow is fairly consistent, and therefore the impacts of the plant can be modeled as a steady forced effort.

An additional source of fish time series data incorporated into the model is an index of biomass generated from the near-shore trawl surveys conducted each fall by the NJDEP. While sampling for this survey occurs along the entire New Jersey coast, it provides an estimate of relative biomass in each year for those species that leave the estuary each fall for offshore or southern waters.

EwE model

The Ecopath model shown in Figure 1 represents a possible configuration of Barnegat Bay for 1981, with the groups arranged by trophic level. There are no surprises in the trophic level of any of the groups, though striped bass in our system do occupy a slightly higher level than those in the Chesapeake Bay. The fact that this model output is parsimonious with other models of similar systems lends additional support to its interpretation. The model is balanced, in that there is sufficient food for the consumers and enough production to meet consumptive demands.

When the time series data is incorporated into the model and the vulnerability values are adjusted to fit to the time series , the overall fit of the model prediction to the available data is reasonable (Figure 2, Sum of Squares = 487.1). The model fits most of the groups well, with changes in relative biomass from the time series data reflected in the model (Figure 3). The increase in relative biomass of croaker throughout the time series is reflective of the increase in its overwintering survivability and general population increase in the Mid-Atlantic as documented by Hare and Able (2007). However the biomass and catch values, particularly the OCNGS catches, appear to be somewhat inflated and warrant further investigation and refinement.

This EcoSim run includes forcing functions for benthic algae and submerged aquatic vegetation (SAV) in an effort to replicate changes in primary producers over time (Figure 4). The benthic algae forcing function is a nearly linear increase from 1981 to 2000 and then no increase for the remainder of the time series, with a 1.5x increase from the beginning to the end of the time series. The SAV function is a steady decrease over the time series to about half of the original. These rates are an estimate of forcing based on the historic decline in SAV and the anecdotal increase in benthic macroalgae.

Figure 1: Barnegat Bay 1981 model. Numbered horizontal lines indicate trophic level.

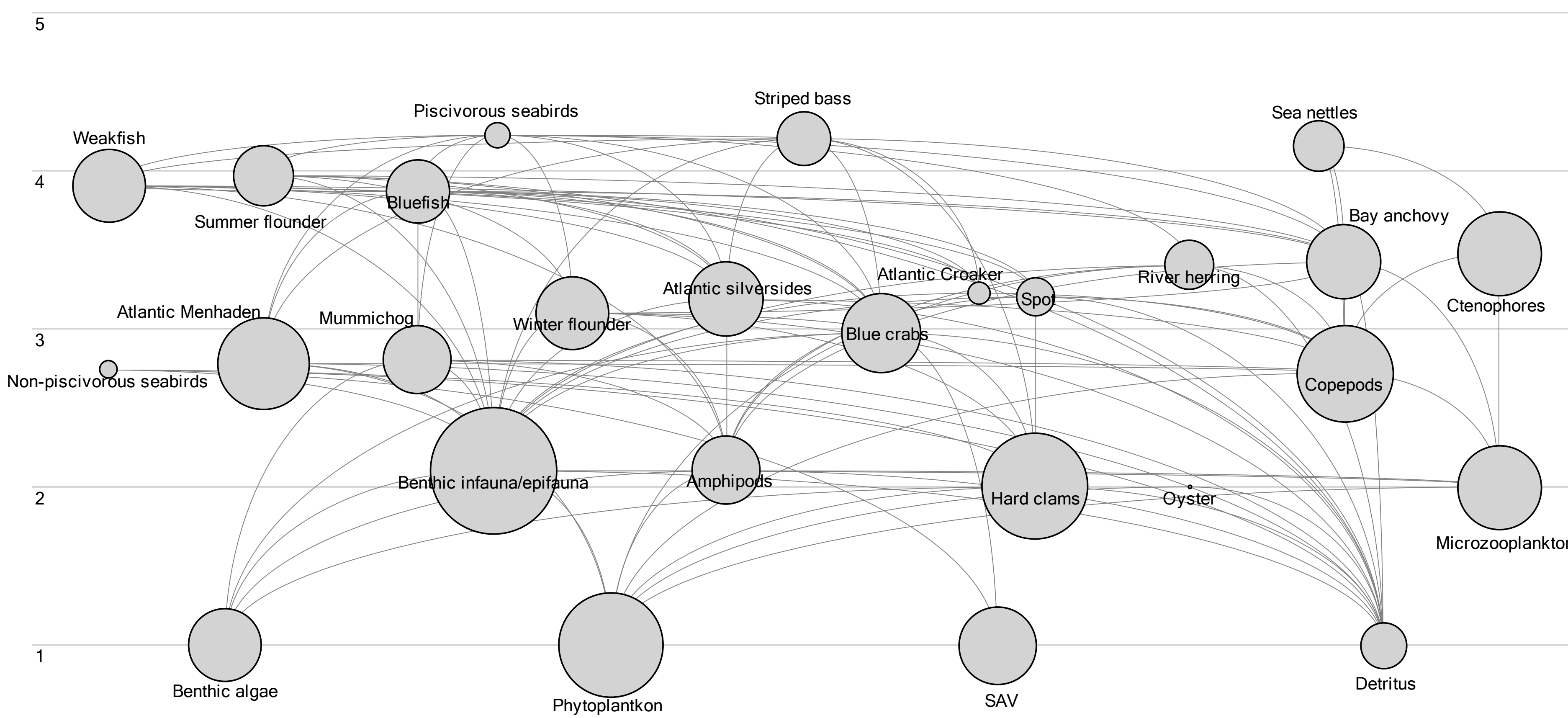
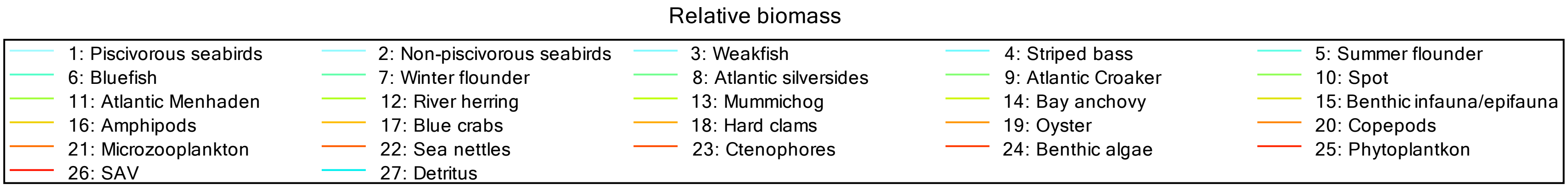


Figure 2: Model predictions versus time series data for 1981 through 2012.



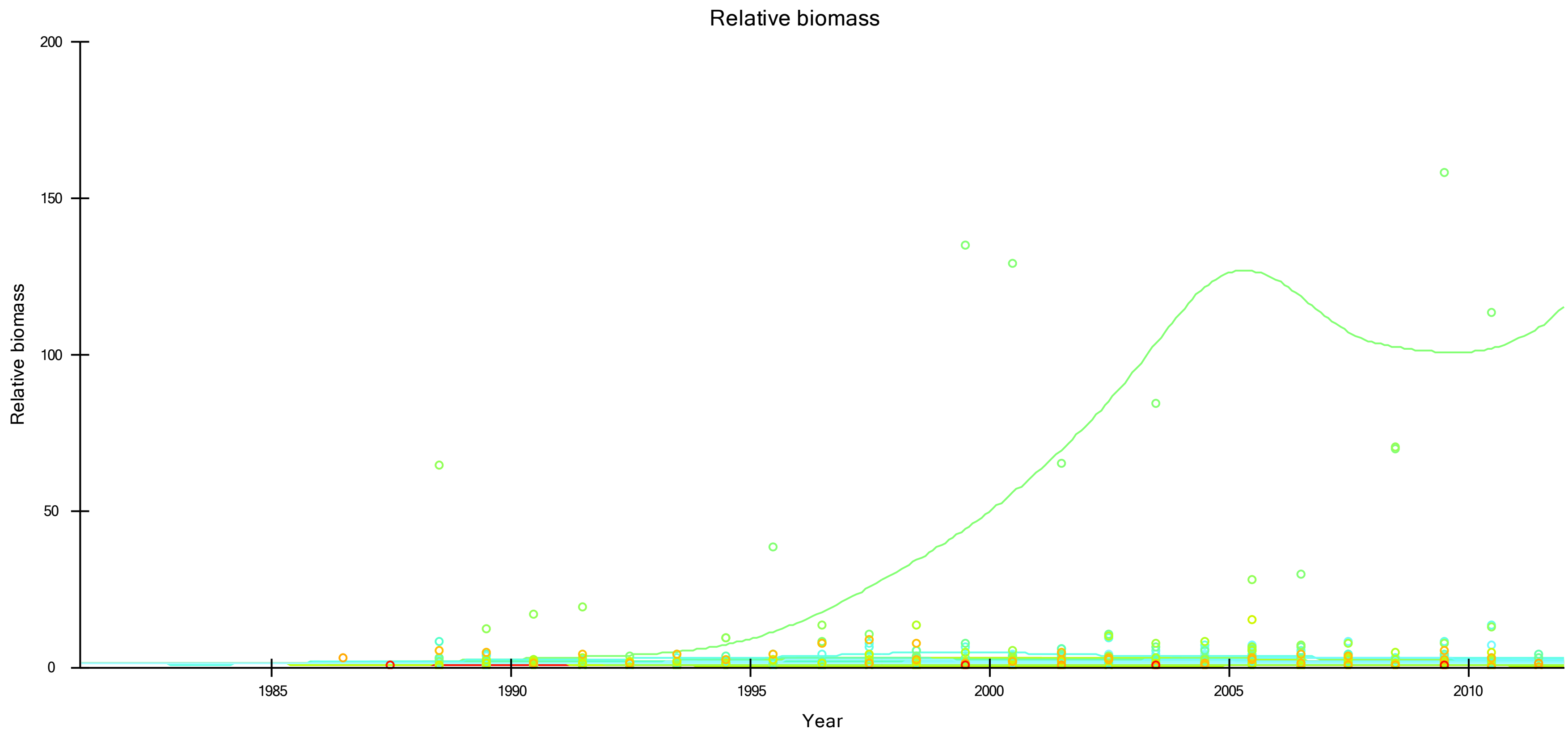

Relative biomass
Relative biomass
200
150
100
50
0
1985
1990
1995
2000
2005
2010
Year

Figure 3: Graphs of the model fit to the currently available time series data for each of the groups in the EwE model.

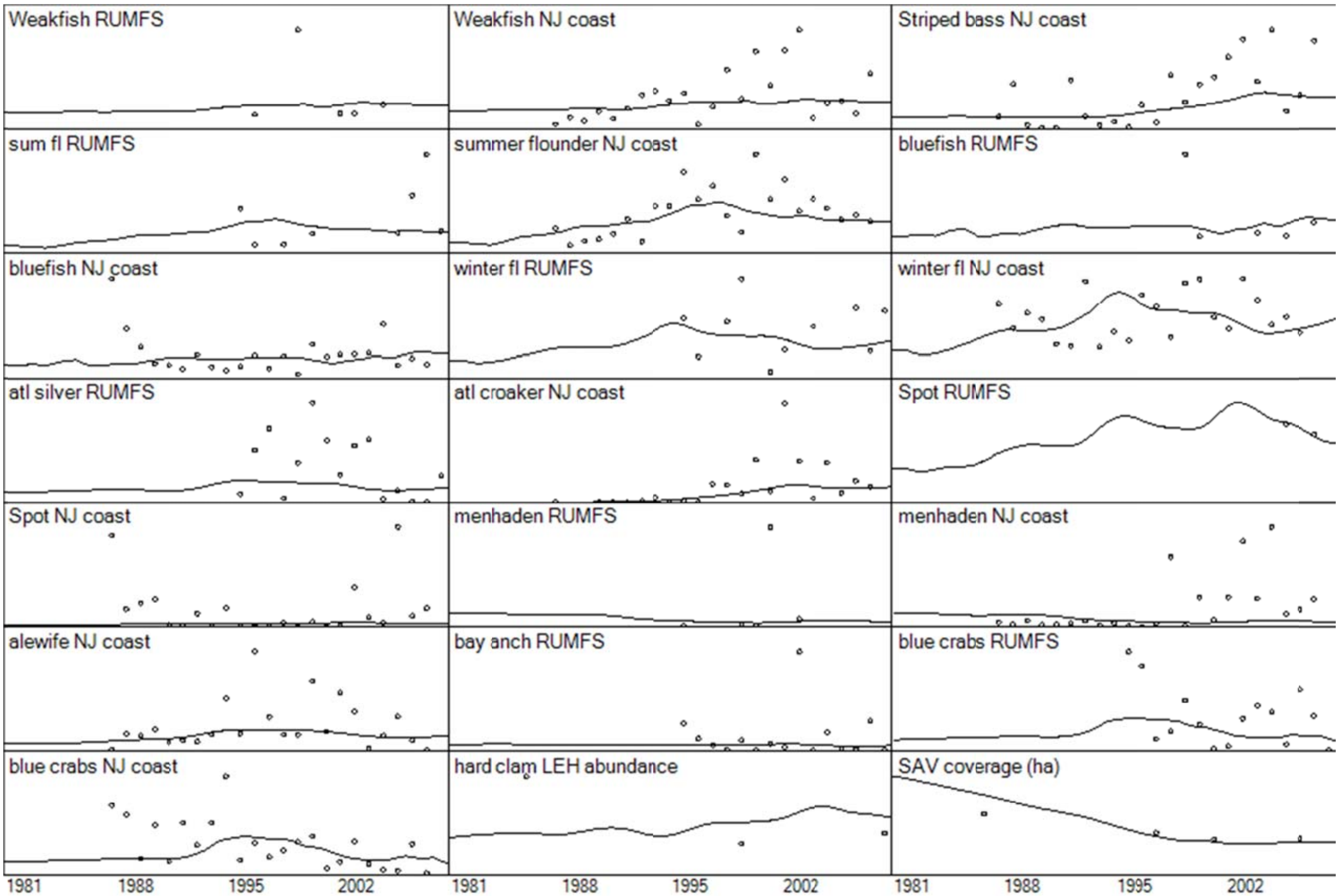
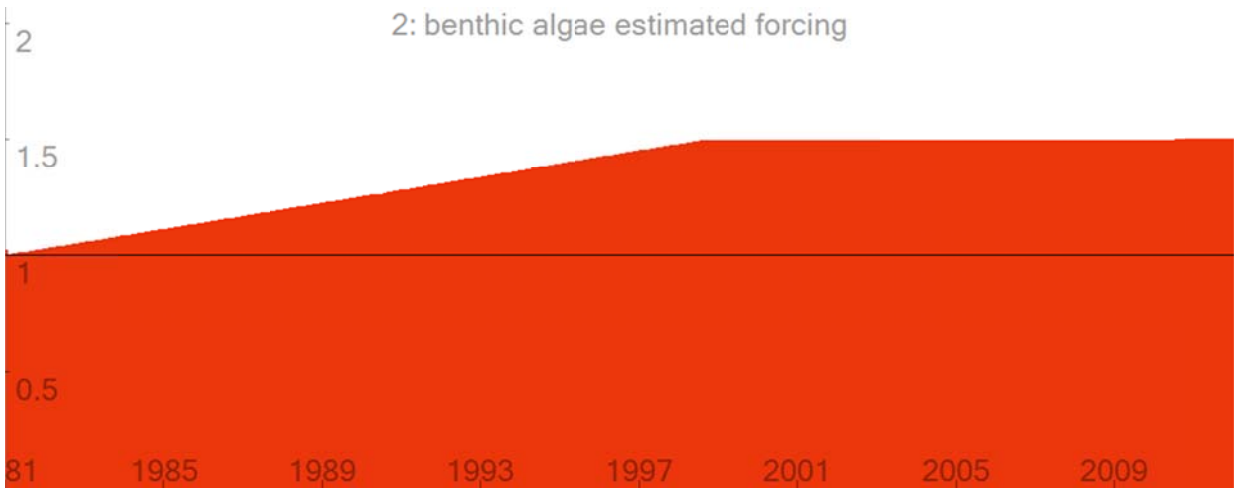
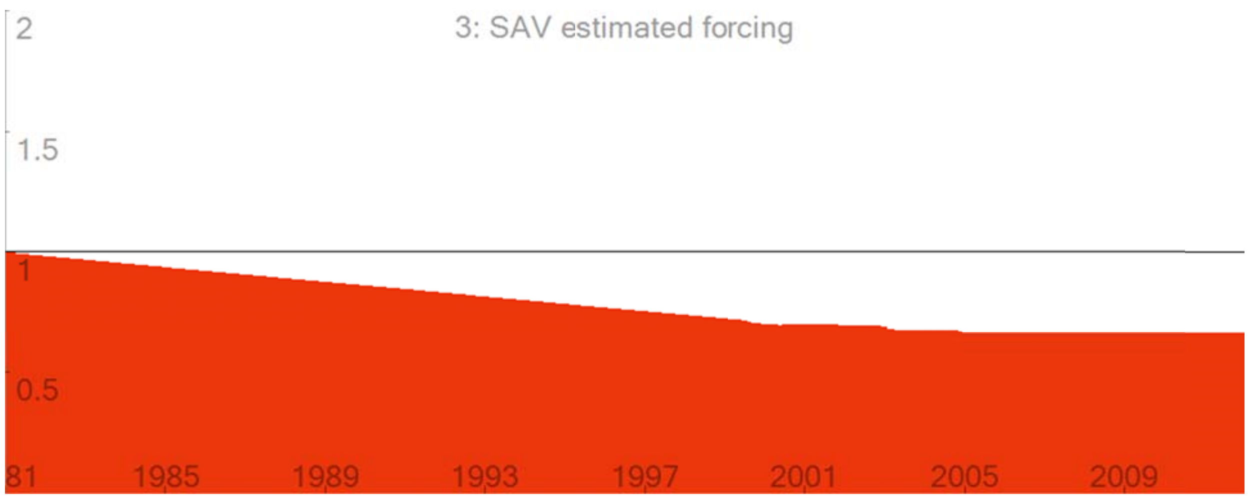


Figure 4: Forcing functions for the EcoSim model run; benthic algae (top), and SAV (bottom).





Nutrient-Phytoplankton-Zooplankton Model

The details and results of the full NPZ model are part of Kevin Crum's Master's thesis "Modeling plankton in a human-impacted estuary: Copepod- vs. jellyfish-dominated Communities", which was approved and accepted by Rutgers University. This thesis formed the basis of a manuscript entitled "Model-data comparisons reveal influence of jellyfish interactions on plankton community dynamics." The manuscript was provided to NJDEP for review on May 12, 2014 and was subsequently submitted to Marine Ecology Progress Series (MEPS), where it was published in December 2014. The published manuscript is attached here as Appendix 5.

Objective 3 - Write and test the program to dynamically link the NPZ and EwE models

The original plan as laid out in the proposal anticipated linking the EwE model to the NPZ model in order to more completely capture the interactions between the lower trophic levels (phytoplankton and zooplankton) and the upper level consumers. This linkage proved to be especially problematic given the different time steps at which the models operate and the internal architecture of the models. While assessing the best way to link the models we were contacted by the USGS Joint Ecological Modeling (USGS-JEM) group to see if they could provide any assistance with data visualization products or model linkages. The USGS received funding to provide assistance to modeling projects within the areas affected by Superstorm Sandy, and they were interested in our project. After a series of emails and conference calls describing our model structure, our needs, and their technical capabilities, we had a meeting March 20-21, 2014 in New Brunswick to outline a plan and timeline for collaboration. At this meeting it was agreed that the USGS-JEM group would build a suite of new visualization tools within the existing EwE software package. Furthermore, the USGS-JEM group will assist in development of a linkage that takes the phytoplankton biomass and production/biomass rates generated by the WASP water quality model being developed by the USGS New Jersey Water Science Center for the Department and pass that information into the EwE model. This model coupling will allow the upper trophic levels of the EwE model to be responsive to changes in nutrients, temperature, or other environmental or management factors that primarily act on lower trophic levels and may not be suitably modeled in EwE. There were some delays in the construction of the WASP model, and therefore this link between WASP and EwE is one of our Year 3 project goals.

However, we have been working with the JEM group to make sure that we will have comparable model groups for when we begin model integration.

Objective 3 Revised – Field assessment of otter trawl efficiency

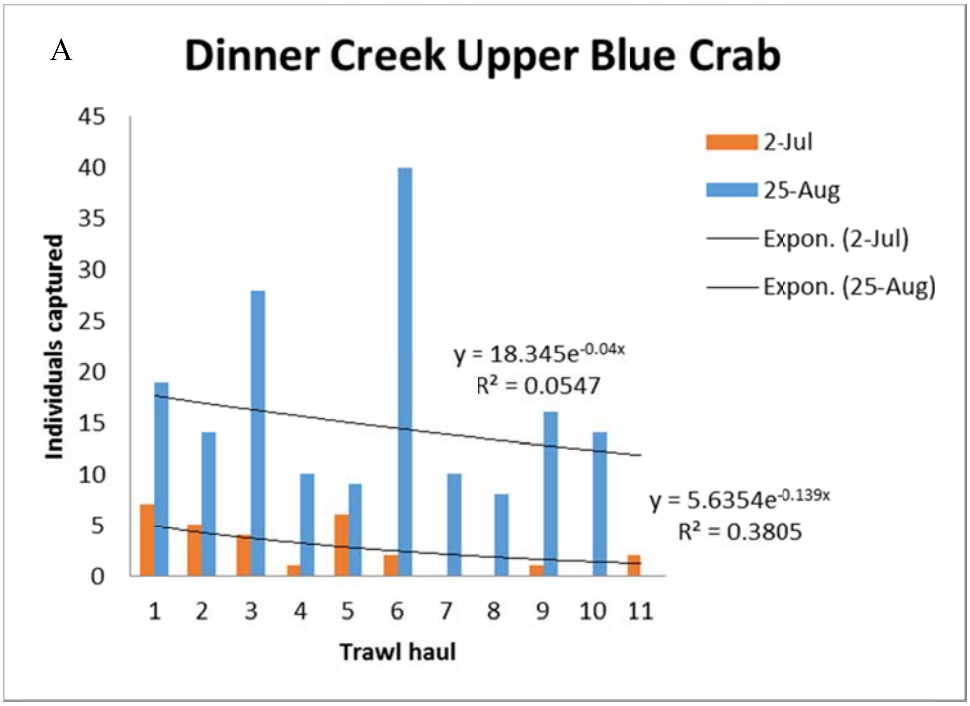
Funds originally allocated for Objective 3 (above) were reallocated toward a field assessment of otter trawl efficiency with the approval of the NJDEP (email from Tom Belton to Olaf Jensen on May 27, 2014).

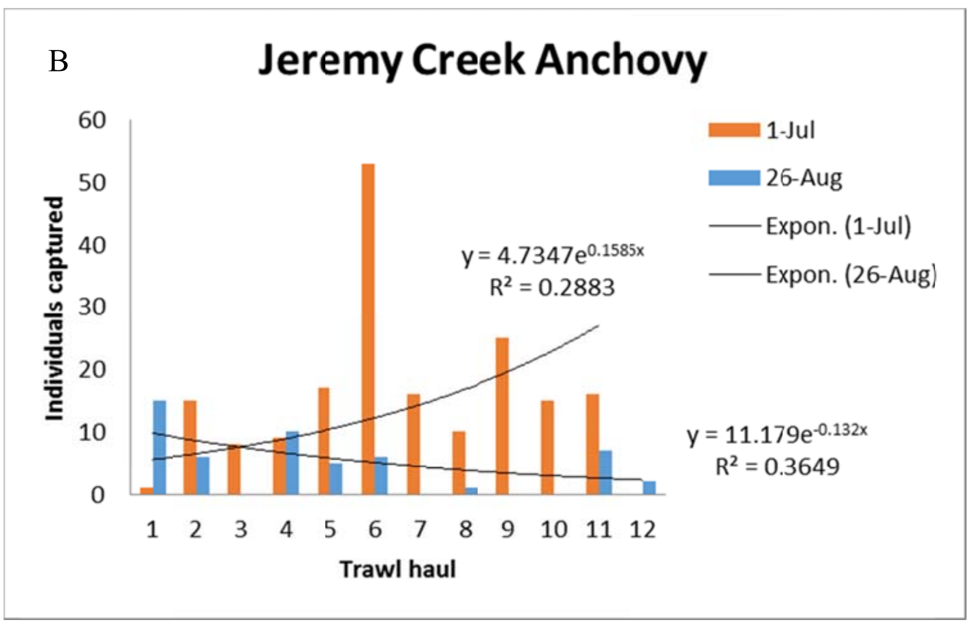
One of the major challenges with development of the EwE ecosystem model is estimating absolute biomass for each of the different trophic groups in Barnegat Bay that are represented in the model. No sampling gear is 100% efficient. That is, all sampling gears capture less than 100% of the organisms encountered. Therefore it is inaccurate to simply estimate biomass based on the number of individuals captured divided by the area or volume sampled. In particular, much of the data on fish and macroinvertebrate abundance used in the EwE model comes from the Rutgers University Marine Field Station's (RUMFS) otter trawl survey.

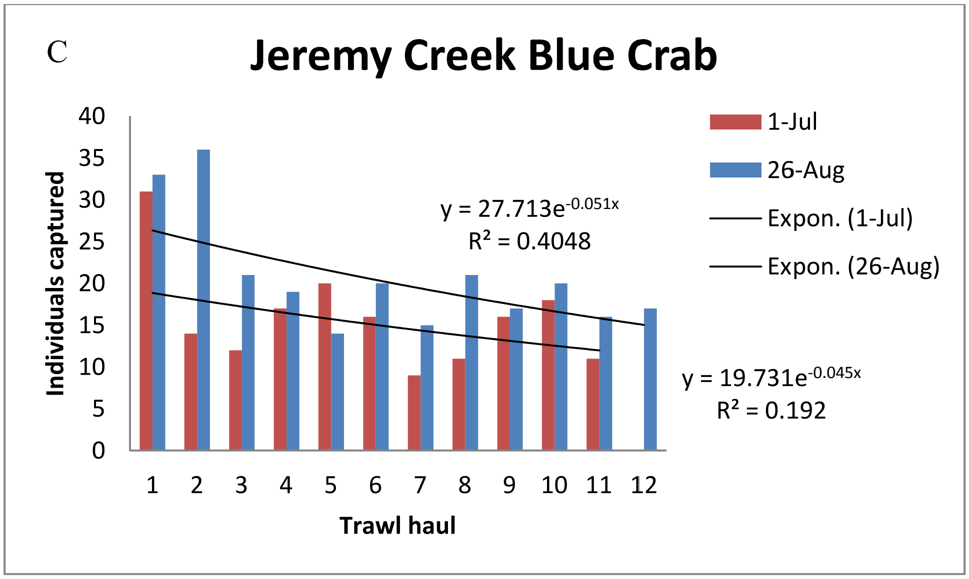
We conducted field assessments of otter trawl efficiency in two sampling events – July 1-3 and August 25-27, 2014 – in three tributaries of Little Egg Harbor. Sites were similar in size, temperature, and salinity to many of the marsh creeks in Barnegat Bay, but were more easily sampled from RUMFS. We set block nets (< 5 mm mesh) across the width of the marsh creek at two locations approximately 50 m apart to isolate the sampled reach from ingress or egress of fish and blue crab. The isolated section of the creek was then repeatedly trawled and all fish and blue crab that were captured were identified to species, recorded, and either removed from the isolated section of the creek (fish) or, for crabs, a leg was clipped at the terminal segment to mark the individual as previously captured and the crab was returned to the isolated section. Catch for the two taxa captured in sufficient numbers (bay anchovy and blue crab) were plotted for each trawl haul and, where appropriate, an exponential curve was fit to the data to estimate the rate of depletion.

There were four site x species combinations, one for bay anchovy and three for blue crab, for which the exponential model was an adequate representation of the observed data (Figure 5). For the other site x species combinations there were either too few individuals captured or no apparent decline in catch. No decline in catch might occur if the trawl efficiency is very low and the abundance of a given species is high or if the block nets did not prevent immigration into the isolated creek section.

If we compare the catch from the first trawl haul to the total catch expected if the isolated creek section were trawled to depletion, we can estimate the trawl efficiency. Trawl efficiency estimated in this manner for blue crab ranged between 4.2% and 22.2% with an average of 11.7%. Efficiency was not estimated for bay anchovy as there was only a single occasion at a single site in which a clear decline in catch was apparent for this species.


A
Dinner Creek Upper Blue Crab
45
40
35
30
25
20
15
10
5
0
Individuals captured
2-Jul
25-Aug
Expon. (2-Jul)
Expon. (25-Aug)
y = 18.345e^-0.04x
R² = 0.0547
y = 5.6354e^-0.139x
R² = 0.3805
1 2 3 4 5 6 7 8 9 10 11
Trawl haul


B
Jeremy Creek Anchovy
60
50
40
30
20
10
0
Individuals captured
1-Jul
26-Aug
Expon. (1-Jul)
Expon. (26-Aug)
y = 4.7347e^0.1585x
R² = 0.2883
y = 11.179e^-0.132x
R² = 0.3649
1 2 3 4 5 6 7 8 9 10 11 12
Trawl haul



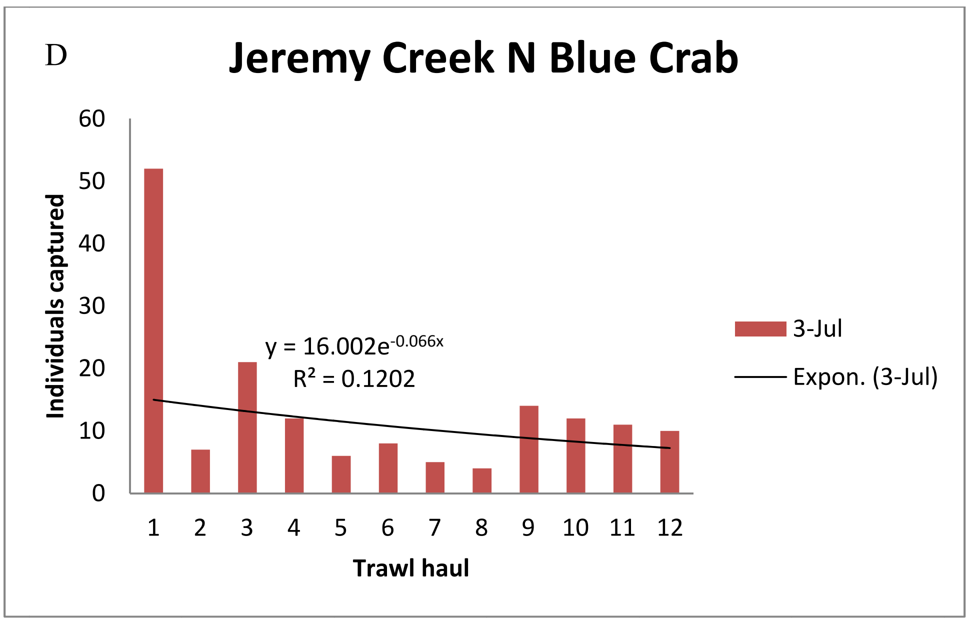


Figure 5. Catch for each trawl haul for blue crab (A, C, and D) and bay anchovy (B) in three locations within tributaries of Little Egg Harbor. Lines represent exponential models fit to the observed catch.

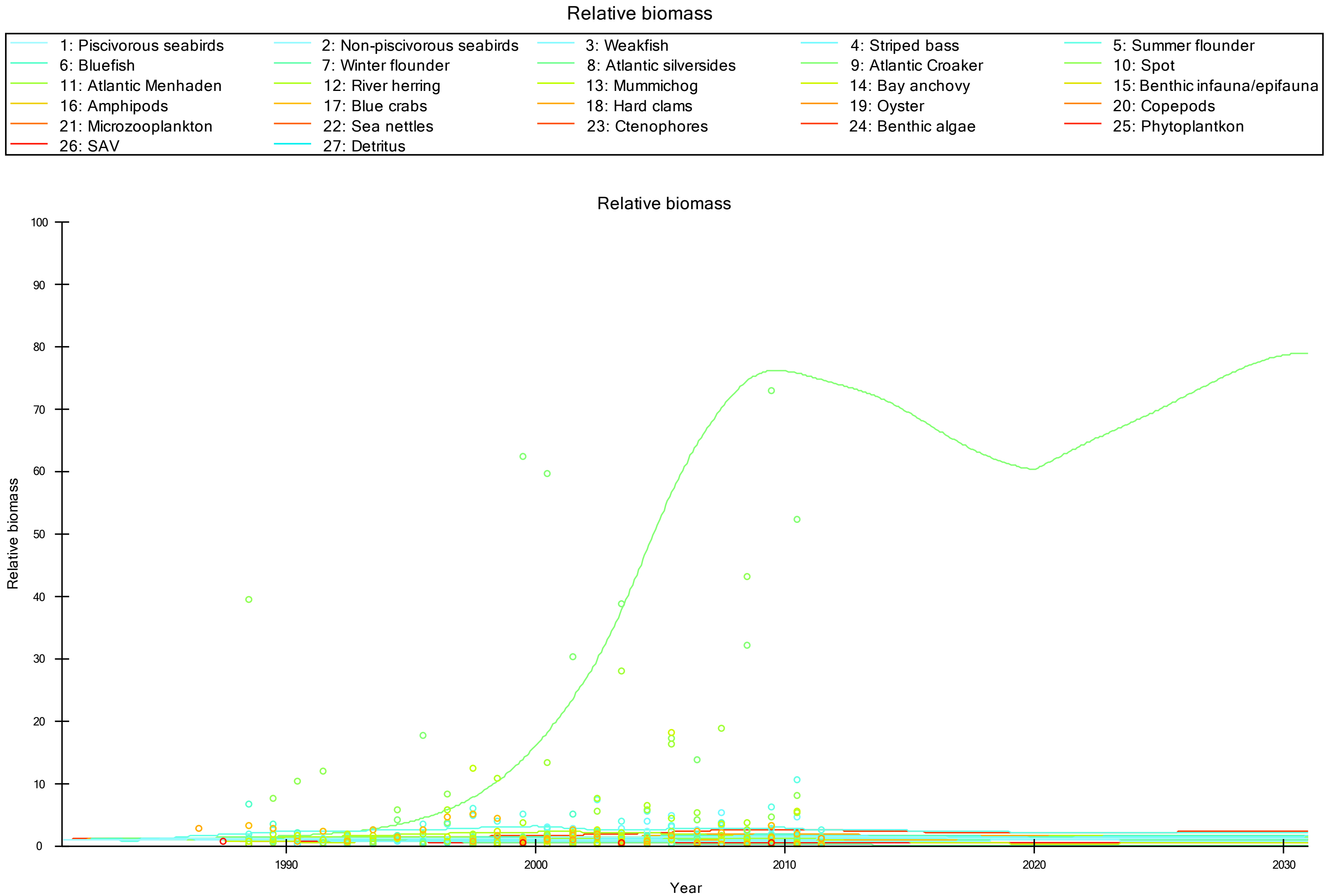
Objective 4 and 5- Develop and run quantitative change scenarios

During the conceptual model development interviews we asked individuals to tell us what variables in their cognitive maps they would increase or decrease the values of in order to effect "positive" change on the bay ecosystem. The responses included changes to both social and ecological components of the system, the impacts of some of which we can model in the EwE, and to a lesser extent in the NPZ, models. We also received input from scientists and managers within the Department regarding what change scenarios they would be most interested in. The following are the results of those changes based on the model as described above.

Scenario 1 – Oyster Creek Nuclear Generating Station (OCNGS) closure

As America's oldest continuously operating nuclear plant, the facility uses a once-through cooling water system, where water is drawn from Forked River, used to cool the plant, and is then returned to Oyster Creek to flow into the bay. The impingement and entrainment of fish, crab, and hard clam larvae, as well as other zooplankters, is well documented. OCNGS functions as a *de facto* fishery, and the removal of biomass from the system is accounted for through catch data used in the EwE model. As part of the Governor's 10-point Plan, the Oyster Creek Nuclear Generating Station (OCNGS) will cease power generation by 2020. To model this scenario we reduced the "catch" of the plant from the "catch" at full operating capacity to 4% of the full operating capacity beginning in 2020, based on the percent reduction in intake water that is planned. The time series data was amended so that the 2011 values for the forced effort series were used for 2012-2030, with the previously noted exception of OCNGS effort. The benthic macroalage, and SAV forcing was set to the 2011 level for the remainder of the simulation. Under those model parameters the relative biomass of most of the groups remains relatively flat or continues along a previous trend, though croaker appears to increase following the plant reduction (Figure 6). If the forced effort data for 2012-2030 are assumed to be the average of the 1981-2011 data the results are similar, though the croaker rebound is dampened slightly.

Figure 6: Model predictions assuming a 96% reduction of OCNGS water uptake from the 1981 value beginning in 2020.



Scenario 2 – Changes to blue crab management strategy

Blue crab are the target of Barnegat Bay's largest commercial fishery, and are currently managed based on a mix of sex and size limits and seasonal closures (NJAC7E:25 and 25A). We modeled the effects of increasing the commercial dredge harvest to 88 metric tons (twice the 1995-2011 average of 44 MT.) and of decreasing the commercial dredge harvest to 22 MT (one-half the ten year average) from 2012 to 2030, while keeping the commercial pot fishery and recreation fisheries at their 1995-2011 averages and the other effort series at their 2011 values. Doubling or halving the commercial dredge had little effect on crab biomass (Figure 7). We also modeled the effects of doubling the commercial pot fishery over the 1995-2011 average of 210 MT to 420 MT and of halving it to 105 MT. Reducing or increasing the landings in the commercial pot fishery had little effect on crab biomass (Figure 8). Even with the commercial pot fishery effort doubled, total catches never exceeded 3MT/km^2, while biomass was predicted to remain steady near 7.5MT/km^2. That is, even a doubling of effort in the commercial pot fishery results in a catch that is too small to have a major impact on the blue crab biomass given estimates of unfished biomass and productivity. We are re-examining estimates of blue crab biomass and productivity using a stock assessment model applied to blue crab landings data.

Scenario 3 – Changes to hard clam management strategy

Hard clams were historically one of the most important commercial fisheries in the Bay, but landings have declined dramatically over the past several decades. We will model the effects of limiting the commercial harvest to 25,000 lbs. (the average of the available landings during the 2000's) during the prediction period (2012-2030) and of closing the fishery entirely for a period of ten years (2012-2022) and then returning to the 25,000 lbs limit. Limiting the commercial harvest to 25,000 lbs. appears to have no effect on hard clam biomass as it fluctuates around 40 t/km2 subsequent to 2011 (Figure 9, left panel). This appears to be primarily driven by natural mortality, which displays a similar pattern. A ten-year moratorium on commercial landings showed identical results (Figure 9, right panel). Both the catch and fishing mortality after 2000 are such a small percentage of the total biomass and total mortality, respectively, that harvest controls have little effect on the population. The large caveats here are that hard clam landings are not recorded by the NJDEP or NMFS, and thus the landing data we obtained appear to be estimates with potentially large uncertainty. Furthermore, the relative paucity of data on hard clams in Barnegat Bay over time made fitting the model particularly difficult for this species.

Figure 7: Changes to the biomass (t/km^2) of blue crab (*Callinectes sapidus*) post 2011 following a doubling of the average dredge fishery effort from 1995-2011 (left panel) and a halving of the effort (right panel).

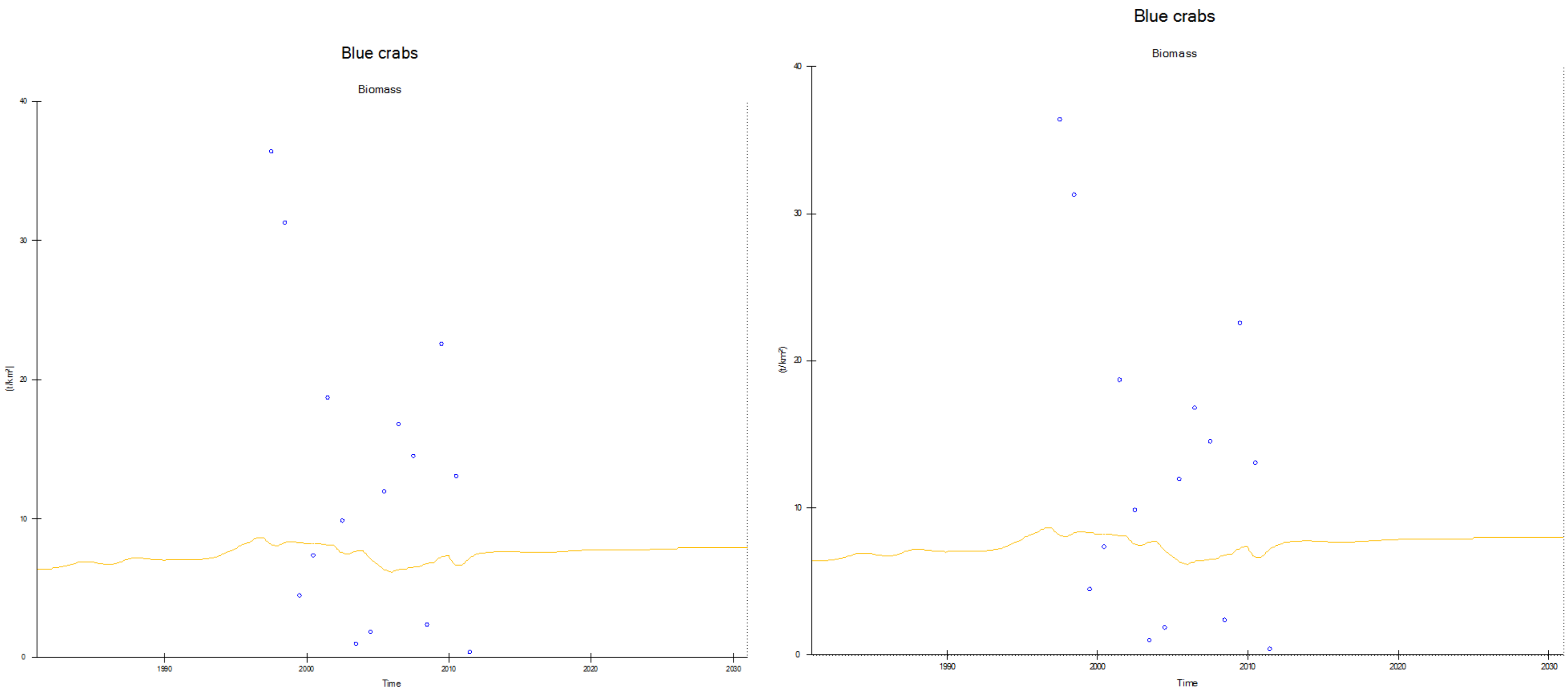


Figure 8: Changes to the biomass (t/km^2) of blue crab (*Callinectes sapidus*) post 2011 following a doubling of the average commercial pot fishery effort from 1995-2011 (left panel) and a halving of the effort (right panel).

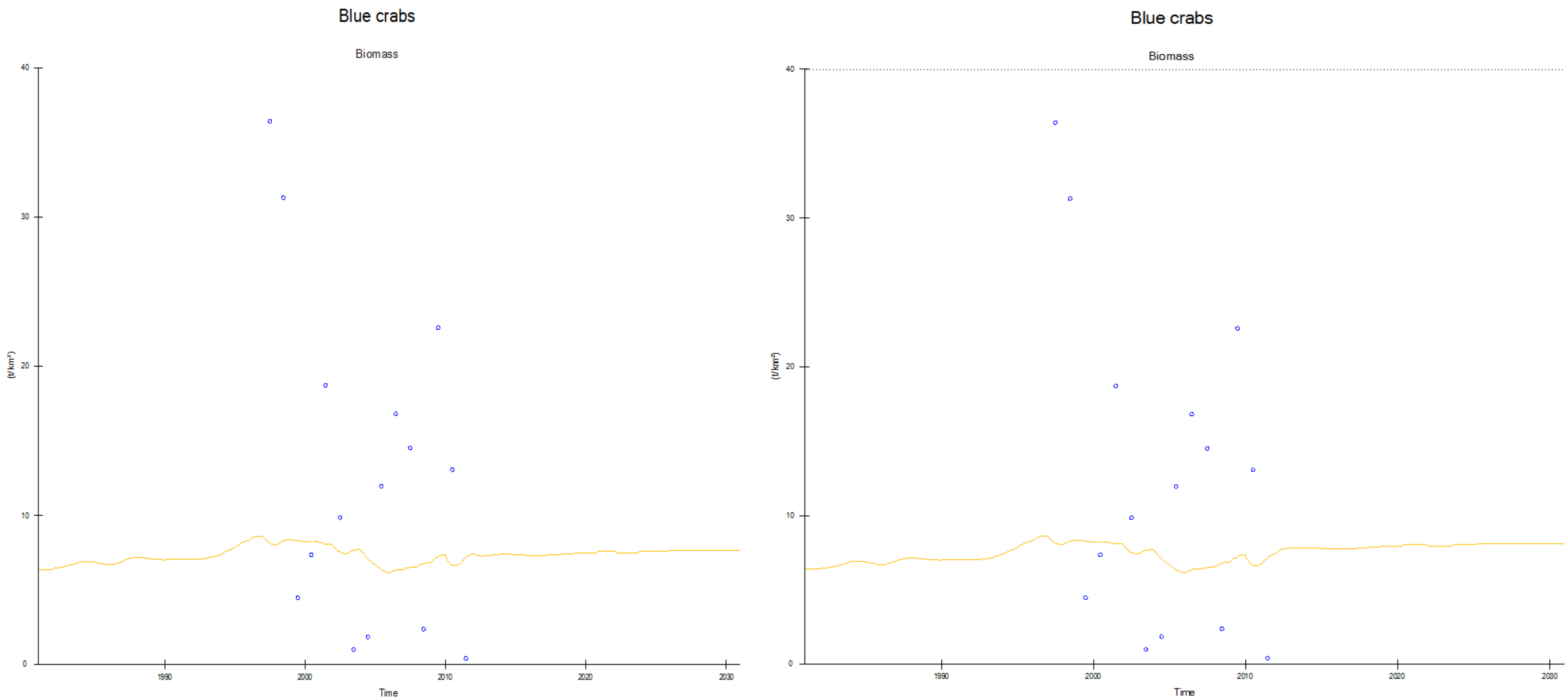


Figure 9: Changes to the biomass (t/km^2) of hard clam (*Mercenaria mercenaria*) post 2011 following a harvest restriction of 25,000 lbs. (left panel) and a closure of the fishery (2012-2012) followed by a limited harvest (right panel).

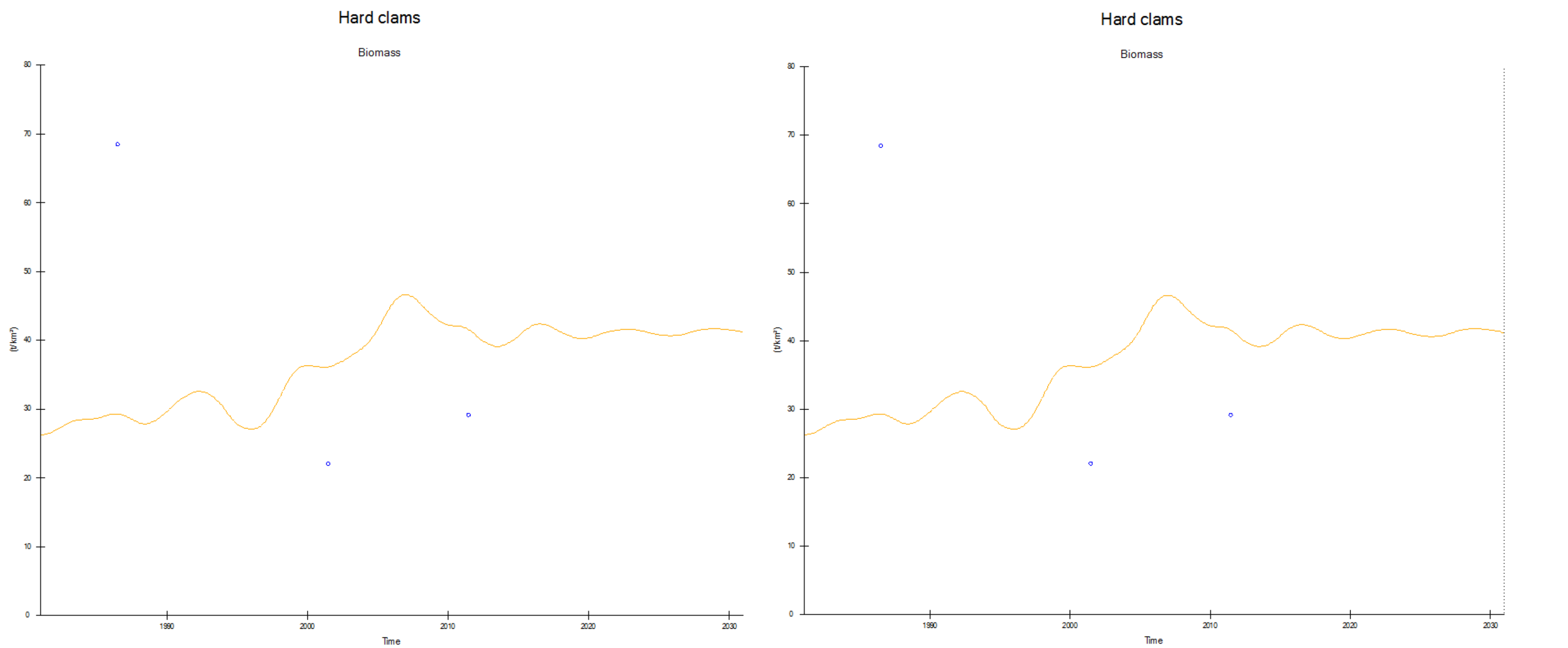
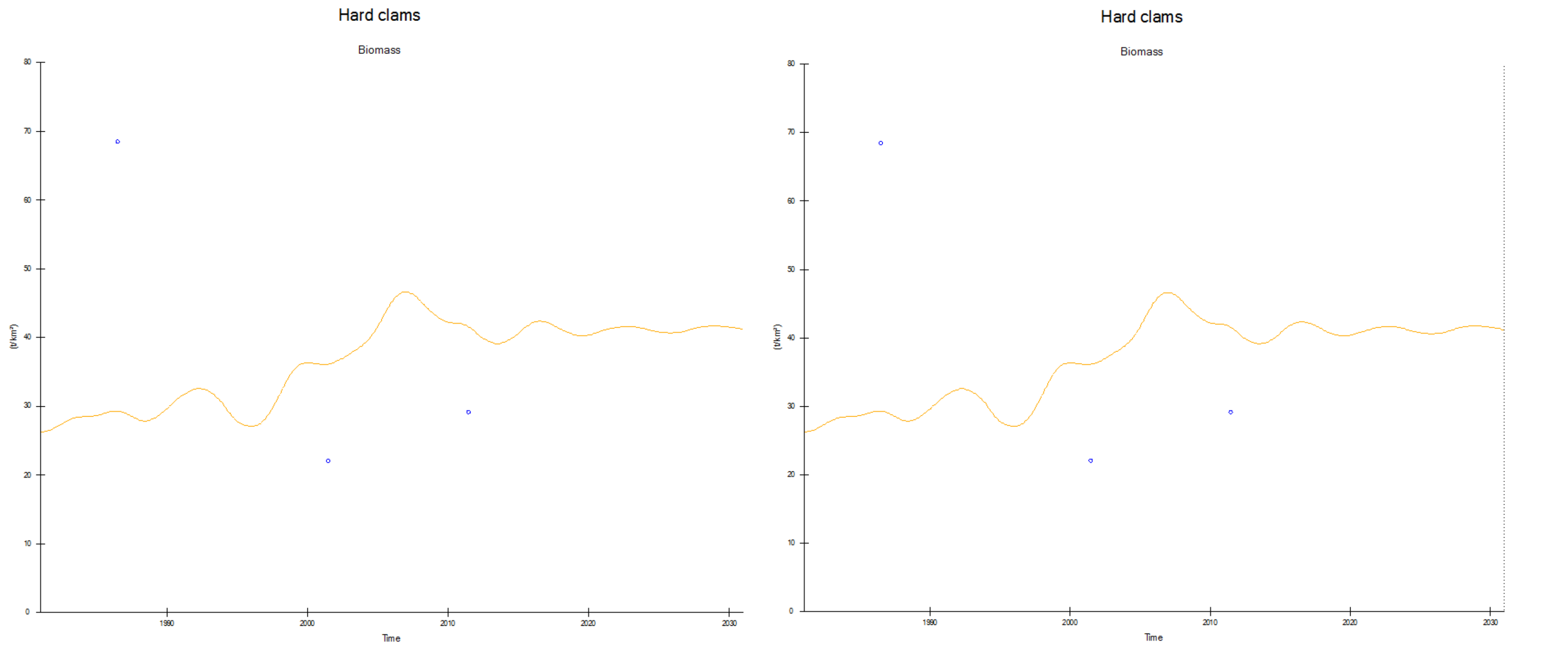


Figure 9: Changes to the biomass (t/km^2) of hard clam (*Mercenaria mercenaria*) post 2011 following a harvest restriction of 25,000 lbs. (left panel) and a closure of the fishery (2012-2012) followed by a limited harvest (right panel).



Scenario 4 – Nutrient input reduction

The Barnegat Bay has been described as a highly eutrophic estuarine system (e.g., Kennish et al. 2007), and the focus of recent legislation (NJ Fertilizer Act, P.L. 2010 Chapter 112; NJ Soil Restoration Act, P.L. 2010 Chapter 113) and restoration efforts (NJ Stormwater Act, P.L. 2010 Chapter 114; Clean Water Act Section 319 projects) in New Jersey has been to reduce the amount of nitrogen being delivered into the system. As no target reductions have been set at this time, we propose to model the effects of reducing nitrogen inputs by 5% and 15%. The effects of these reductions will be felt most directly on phytoplankton and zooplankton biomass, and can be most appropriately modeled in the WASP model that is currently under production. Once the linkage between the WASP and EwE model is constructed we will be able to pass the changes along to the upper trophic levels.

References

Anderson, D.R., 1975. Population ecology of the mallard, V: Temporal and geographic estimates of survival, recovery, and harvest rates. U.S. Fish and Wildl. Serv. Resour. Publ., 25:110.

ASMFC. 2003 Atlantic striped bass advisory report. ASMFC Striped Bass Technical Committee Report 2003-03, Atlantic States Marine Fisheries Commission, Washington, D.C.

Baird, D. and Ulanowicz, R.E., 1989. The seasonal dynamics of the Chesapeake Bay ecosystem. Ecological Monographs, 59:329-364.

Bougon, M., Weick, K., Binkhorst, D., 1977. Cognition in organizations: an analysis of the Utrecht Jazz Orchestra. Admin. Sci. Quart. 22, 606–639.

Bricelj, V. M., J.N. Kraeuter, and G. Flimin. 2013. Status and Trends of Hard Clam, *Mercenaria mercenaria*, Shellfish Populations in Barnegat Bay, New Jersey. Barnegat Bay Partnership Technical Report. Toms River, Barnegat Bay Partnership**:** 143.

Christensen, V. and Walters, C.J., 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecol. Model., 172:109-139.

Christensen, Villy, and Alasdair Beattie, Claire Buchanan, Hongguang Ma, Steven J. D. Martell, Robert J. Latour, Dave Preikshot, Madeline B. Sigrist, James H. Uphoff, Carl J. Walters, Robert J. Wood, and Howard Townsend. 2009. Fisheries Ecosystem Model of the Chesapeake Bay: Methodology, Parameterization, and Model Explanation. U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-F/SPO-106, 146 p.

Christensen, V., Walters, C.J. and Pauly, D., 2005. Ecopath with Ecosim: a User's Guide, November 2005 Edition, Fisheries Centre, University of British Columbia, Vancouver, Canada.

Carley, K., Palmquist, M., 1992. Extracting, representing, and analyzing mental models. Social Forces 70, 601–636.

Eden, C., Ackerman, F., Cropper, S., 1992. The analysis of cause maps. J. Manage. Stud. 29, 309–323

Frisk, M.G., T.J. Miller, R.J. Latour, and S. Martell. 2006. An ecosystem model of Delaware Bay.

Froese, R. and Pauly, D., 2004. FishBase, World Wide Web electronic publication, www.fishbase.org,version (03/2013).

Fuchs HL, Franks PJS (2010). Plankton community properties determined by nutrients and size selective feeding. Marine Ecology Progress Series, 413: 1-15.

Gray, S., A. Chan, D. Clark, R. Jordan. 2011. Modeling the integration of stakeholder knowledge in social–ecological decision-making: Benefits and limitations to knowledge diversity. Ecol. Model. doi:10.1016/j.ecolmodel.2011.09.011

Hage, P., Harary, F., 1983. Structural Models in Anthropology. Oxford University Press, New York.

Harary, F., Norman, R.Z., Cartwright, D., 1965. Structural Models: An Introduction to the Theory of Directed Graphs. John Wiley & Sons, New York.

Hobbs, B.F., Ludsin, S.A., Knight, R.L., Ryan, P.A., Biberhofer, J., Ciborowski, J.J.H., 2002. Fuzzy cognitive mapping as a tool to define management objectives for complex ecosystems. Ecol. Appl. 12, 1548–1565.

Houde, E.D. and Zastrow, C.E., 1991. Bay anchovy (*Anchoa mitchilli*). In: S.L. Funderburk, J.A. Mihursky, S.J. Jordon and D. Riley (Editor), Habitat requirements for Chesapeake Bay living resources. 2nd edition. Chesapeake Bay Program Office, U.S. Environmental Protection Agency, Annapolis, Md., pp. 8:1-14.

Hoenig, J. M. 1983. Empirical Use of Longevity Data to Estimate Mortality-Rates. Fishery Bulletin **81**:898-903.

ICES, 2000. Report of the working group on seabird ecology, ICES CM 2000/C:04

Jørgensen, L.A., Jørgensen, S.E. and Nielsen, S.N., 2000. ECOTOX: Ecological Modelling and Ecotoxicology. Elsevier Science B.V., Amsterdam.00

Kahn, D. M. 2003. Stock assessment of Delaware Bay blue crab (Callinectes sapidus) for 2003. Div. Fish Wild., Dover, DE.

Kahn, D. M., and Helser T. E. 2005. Abundance dynamics and mortality rates of the Delaware Bay stock of blue crabs, Callinectes sapidus. Journal of Shellfish Research **24**:269-284.

Kennish, M.J. 2001. The Scientific Characterization of the Barnegat Bay – Little Egg Harbor Estuary and Watershed. Jacques Cousteau National Estuarine Research Reserve Contribution #100-5-01.

Kennish MJ (2001a). Physical description of the Barnegat Bay—Little Egg Harbor estuarine system. Journal of Coastal Research, SI(32): 13-27.

Kennish, M.J., S.B. Bricker, W.C. Dennison, P.M. Glibert, R.J. Livingston, K.A. Moore, R.T. Noble, H.W. Paerl, J.M. Ramstack, S. Seitzinger, D.A. Tomasko, and I. Valiela. 2007. Barnegat Bay–Little Egg Harbor Estuary: case study of a highly eutrophic coastal bay system. Ecological Applications 17: S3–S16.

Kennish, M.J., B.M. Ferting, G.P. Sakowicz. 2013. In situ Surveys of Seagrass Habitat in the Northern Segment of the Barnegat Bay - Little Egg Harbor Estuary: Eutrophication Assessment. Barnegat Bay Partnership Technical Report. 43p.

Kim, H.S., Lee, K.C., 1998. Fuzzy implications of fuzzy cognitive map with emphasis on fuzzy causal relationships and fuzzy partially causal relationship. Fuzzy Sets Syst. 97, 303–313.

Kosko, B., 1986. Fuzzy Cognitive Maps. Int. J. Man–Machine Stud. 24, 65–74.

Lathrop, R. G. , R.M. Styles, S. P. Seitzinger, J.A. Bognar. 2001. Use of GIS Mapping and Modeling Approaches to Examine the Spatial Distribution of Seagrasses in Barnegat Bay, New Jersey. Estuaries 24(6A): 904-916.

Lowerre-barbieri, S. K., Chittenden M. E., and Barbieri L. R. 1995. Age and Growth of Weakfish, Cynoscion Regalis, in the Chesapeake Bay-Region with a Discussion of Historical Changes in Maximum Size. Fishery Bulletin **93**:643-656.

Luo, J. and Brandt, S.B., 1993. Bay anchovy, Anchoa mitchilli, production and consumption in mid-Chesapeake Bay based on a bioenergetics model and acoustic measurement of fish abundance. Marine Ecology Progress Series, 98:223-236.

Macro International Inc. 2008. New Jersey Blue Crab Recreational Fishery Survey 2007 Final Report.

Matishov, G.G. and Denisov, V.V., 1999. Ecosystems and biological resources of Russian European seas at the turn of the 21st century, Murmansk Marine Biological Institute, Murmansk

Moser FC (1997). Sources and sinks of nitrogen and trace metals, and benthic macrofauna assembleges in Barnegay Bay, New Jersey. PhD Dissertation. Rutgers University, New Brunswick, New Jersey, USA.

Nemerson, D. M., and Able K. W. 2004. Spatial patterns in diet and distribution of juveniles of four fish species in Delaware Bay marsh creeks: factors influencing fish abundance. Marine Ecology-Progress Series **276**:249-262.

Olsen PS, Mahoney JB (2001). Phytoplankton in the Barnegat Bay-Little Egg Harbor estuarine system: Species composition and picoplankton bloom development. Journal of Coastal Research, SI(32): 115-143.

Oshima, Y., Kishi, M.J. and Sugimoto, T., 1999. Evaluation of the nutrient budget in a seagrass bed. Ecol Model, 115:19-33.

Özesmi, U., Özesmi, S., 2003. A participatory approach to ecosystem conservation: fuzzy cognitive maps and stakeholder group analysis in Uluabat Lake, Turkey. Environ. Manage. 31 (4), 518–531.

Özesmi, U., Özesmi, S., 2004. Ecological models based on people's knowledge: a multi-step fuzzy cognitive mapping approach. Ecol Model. 176:43-64.

Palomares, M. L. D. 1991. La consummation de nourriture chez les poissons: étude comparative, mise au point d'un modèle prédictif et application à l'étude des réseaux trophiques. Thèse de Doctorat, Institut National Polytechnique de Toulouse:211.

Palomares, M.L.D. and Pauly, D., 1998. Predicting food consumption of fish populations as functions of mortality, food type, morphometrics, temperature and salinity. Mar. Freshwat. Res., 49:447-453.

Park, G.S. and Marshall, H.G., 2000. The trophic contributions of rotifers in tidal freshwater and estuarine environments. Estuarine, Coastal and Shelf Science, 51:729-742.

Pauly, D. 1989. Food consumption by tropical and temperate fish populations: some generalizations. J. Fish Biol. **35(Suppl. A)**:11-20

Piner, K. R., and Jones C. M. 2004. Age, growth and the potential for growth overfishing of spot (Leiostomus xanthurus) from the Chesapeake Bay, eastern USA. Marine and Freshwater Research **55**:553-560.

Preikshot, D., 2007. The influence of geographic scale, climate and trophic dynamics upon North Pacific oceanic ecosystem models. Ph.D. , University of British Columbia, Vancouver

Randall, R.G. and Minns, C.K., 2000. Use of fish production per unit biomass ratios for measuring the productive capacity of fish habitats. Canadian Journal of Fisheries and Aquatic Sciences, 57:1657-1667.

Ross, S. W. 1988. Age, growth, and mortality of Atlantic croaker in North Carolina, with comments on population dynamics. Trans. Am. Fish. Soc. **117**:461-473.

Sellner, K.G., Fisher, N., Hager, C.H., Walter , J.F. and Latour, R.J., 2001. Ecopath with Ecosim Workshop, Patuxent Wildlife Center, October 22-24, 2001, Chesapeake Research Consortium, Edgewater MD

Shushkina, E.A., Musaeva, E.I., Anokhina, L.L. and Lukasheva, T.A., 2000. The role of gelatinous macroplankton, jellyfish *Aurelia*, and Ctenophores *Mnemiopsis* and *Beroe* in the planktonic communities of the Black Sea. Russian Academy of Sciences. Oceanology, 40:809-816.

Sissenwine, M., 1987. Chapter 31. Fish and squid production. In: R.H. Backus and D.W. Bourne (Editor), Georges Bank. MIT Press, Cambridge, Mass., pp. 347-350.

Smith, D.R., Burnham, K.P., Kahn, D.M., He, X. and Goshorn, C.J., 2000. Bias in survival estimates from tag-recovery models where catch-and-release is common, with an example from Atlantic striped bass. Canadian Journal of Fisheries and Aquatic Sciences, 57:886-997

Sugihara, T., C. Yearsley, J.B. Durand, N.P. Psuty. 1979. Comparison of Natural and Altered Estuarine Systems. Center for Coastal and Environmental Studies, Rutgers – The State University of New Jersey. CCES Publication NJ/RU – DEP-11-9-79.

Tomasko, D. A., C. J. Dawes, M.O. Hall. 1996. "The effects of anthropogenic nutrient enrichment on turtle grass (Thalassia testudinum) in Sarasota Bay, Florida." Estuaries **19**(2B): 448-456.

Appendix 1 – Ecopath Parameter Derivations

Fish

Atlantic Croaker

Q/B - Estimates of consumption to biomass ratio was calculated in FishBase as 4.2 year^{-1}, assuming an annual temperature of the Barnegat Bay of T = 15 oC, aspect ratio = 1.32, W*inf* = 815.3, and carnivorous feeding.

P/B - An annual total mortality for the Chesapeake Bay Atlantic croaker stock was estimated to be 55 to 60% per year (Austin *et al*., 2003). Using the higher end as a conservative mortality estimate yields a P/B = 0.916 year^{-1}.

Biomass – An EE value of 0.90 was used and EwE estimated the biomass. Croaker were rarely identified in the Sugihara et. al (1979) study and thus the Delaware Bay and Chesapeake models likely overestimate the biomass present here.

Diet – The diet data is based on the general diet found in the Delaware Bay model, which is a composite of the Nemerson and Able (1994) study.

Atlantic Menhaden

Q/B – A value of 31.42 year^{-1} taken from Palomares and Pauly (1998).

P/B – As there was no commercial fishery for menhaden in Barnegat Bay and only a limited bait fishery, total mortality was set equal to natural mortality, which is estimated at 0.50 year^{-1} (MSVPA-X averaged across all ages and 1982-2008; in 2010 Stock Assessment Table 2.13).

Biomass – Biomass was calculated by EwE setting the EE to 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Atlantic Silverside

Q/B – The consumption ratio for silversides of 4.0 year^{-1} was determined by setting a production/consumption ratio of 0.2 (Christensen et al.).

P/B – Total mortality for littoral forage fish was estimated by local experts at a Chesapeake Bay Ecopath Workshop (Sellner *et al.*, 2001) to be 0.8 year^{-1}.

Biomass - The biomass for the group was estimated by setting ecotrophic efficiency to 0.95. While baywide biomass was not determined by Vougliotis et al (1987), they suggested it should be comparable, if not great than what they determined for bay anchovy, given Atlantic silverside was numerically dominant.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Bay Anchovy

Q/B - Assuming habitat temperature of 15 ^{o}C, W∞ = 20 (g), an aspect ratio of 1.32, and carnivorous diet, the consumption to biomass ratio is calculated by Fishbase to be 9.7 year^{-1}.

P/B – Christensen et al used an initial P/B of 3.0 year^{-1} for the Chesapeake Bay model based on a 95% annual mortality rate reported by Luo and Brandt (1993), while Frisk et al. (2006) estimated a P/B of 2.19 year^{-1} from catch curve analysis on adults in Delaware Bay. We elected to use the higher rate.

Biomass – Vougliotis et al (1987) estimated biomass for 1976 to range from 0.83 to 4.83 g/m^2. In the same study the catch per unit effort for 1981 was comparable to that for 1976, and thus the biomass range should be similar. Given the ubiquity of the species within the Barnegat Bay, I chose to use $4.83 g/m^2$ for an initial biomass.

Diet - Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Bluefish

Q/B - Assuming habitat temperature of 15 $_o$C, W_{max} = 16,962.1 (g), carnivorous feeding, and an aspect ratio of 2.55, the resulting consumption to biomass ratio is 3.1 $year^{-1}$.

P/B – Production/biomass was determined as 0.52 $year^{-1}$ based on an M = 0.25 $year^{-1}$ (Christensen et al) and an estimate of F = 0.27 $year^{-1}$ for 1982 from the 41st Stock Assessment Workshop (2005) for Bluefish (Figure B2).

Biomass – Biomass was calculated by EwE setting the EE to 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study averaged for all size classes.

Mummichog

Q/B – A Q/B of 3.65 $year^{-1}$ was used (Pauly1989).

P/B – We opted to utilize a P/B of 1.2 $year^{-1}$ as given in Frisk et al (2006) from “best professional judgement” compared to Valiela 0.287 $year^{-1}$ (1977 mortality tables) or Christensen et al’s 0.8 $year^{-1}$.

Biomass- The biomass for the group was estimated by setting ecotrophic efficiency to 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

River herring

Q/B – We used a Q/B = 8.4 $year^{-1}$, which is the average of Pauly (1989; 8.63 at temperature = 10C) and Palomares (1991; 8.23 at temperature= 20C).

P/B - Total mortality for this group was based on the P/B of 0.75 $year^{-1}$ for alewife in Randall and Minns (2000).

Biomass – Biomass was estimated by EcoPath assuming that the ecotrophic efficiency of these species in the Bay was 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Spot

Q/B – The consumption biomass ratio was estimated as 6.2 $year^{-1}$ using the model in Fishbase.org and a habitat temperature of 15 $_0$C, W_∞ = 190g (Piner and Jones, 2004) and an aspect ratio of 1.39 (Christensen et al).

P/B - Hoenig’s method estimated an M = 0.9 $year^{-1}$ given a maximum age of 5 (Piner and Jones, 2004). This is consistent with the Z used in the Delaware Bay model.

Biomass – Biomass was estimated by the software using an EE value of 0.90, which was taken from the Chesapeake Bay model.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Striped bass

Q/B - Based on empirical relationship provided by Fishbase.org and assuming an aspect ratio of 2.31 (Chesapeake Bay Ecopath Model), temperature T = 15 ^{0}C, and W_∞ = 46.6 kg (Funderbunk et al 1991), the estimated consumption ratio was 2.4 year^{-1}.

P/B – The 1981 ASMFC FMP suggest an M=.15 and an F=.3 for the coastwide stock. Given the reduced fishing mortality in the Barnegat Bay, an F=.25 is appropriate leading to a P/B of 0.4 year^{-1}. This is equal to the Chesapeake model for resident bass (1-7 years old), though their YOY P/B = 1.8 year^{-1}.

Biomass – The biomass was estimated by EcoPath based on an EE of 0.90.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study and was averaged across all size classes.

Summer Flounder

Q/B- Assuming an aspect ratio of 1.32, W_{max} = 12kg (Frisk et al 2006), carnivorous feeding, and habitat temperature of 15 °C, the consumption to biomass ratio is = 2.6 year^{-1}.

P/B- The Chesapeake Bay and Delaware Bay models utilized P/B=0.52 year^{-1} based on the 2002 NEFSC determination of M=0.2 and F ranging between 0.24 and 0.32.

Biomass – Biomass was estimated by the software using an EE value of 0.95, which is in-line with that used in the Chesapeake Bay model.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Weakfish

Q/B - Using Fishbase, consumption to biomass was estimated = 3.0 year^{-1}, assuming average habitat temperature of 15 °C, aspect ratio of 1.32, maximum weight W_∞ = 6,190g (Lowerre-Barbieri et al., 1995) and carnivorous feeding habitats.

P/B –Total mortality of Z = 0.26 year^{-1} was estimated using Hoenig's method (1983) assuming a longevity of 17 years (Lowerre-Barbieri et al., 1995). This is in-line with an estimated M of .25 year^{-1} as used for stock assessment purposes (Smith *et al.*,2000). Given the low rate of fishing in Barnegat Bay, Hoenig's estimation of Z seem reasonable.

Biomass – Biomass was estimated by the software using an EE value of 0.90.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study and averaged across all size classes.

Winter Flounder

Q/B - The estimated consumption ratio of 3.4 year^{-1} was derived using the empirical equation in FishBase (Froese and Pauly, 2004), and was calculated assuming that T = 15 °C, W_{inf} = 3,600 g (Fishbase), an aspect ratio of 1.32, and a carnivorous diet.

P/B – The 2011 Southern New England/Mid-Atlantic stock assessment updated natural mortality (M) to 0.30 year^{-1} for all ages and all years. Fishing mortality for ages 4-6 was determined as 0.61 year^{-1} for 1981. If one assumes only natural mortality for ages 0-3 and then F+M for ages 4-6, total mortality (Z) is 0.52 averaged across all ages.

Biomass – Biomass was estimated by the software using an EE value of 0.95.

Diet – Diet data is from the Rutgers University 1979 Manahawkin Bay study.

Piscivorous seabirds

Q/B - The consumption ratio estimate of 120 $year^{-1}$ was from data for the piscivorous seabirds group in Preikshot (2007).

P/B - A total mortality estimate for piscivorous seabirds of 0.163 $year^{-1}$ was based on survival rate values of 85-90% for cormorants and 80-93% for alcids in the northeast Atlantic (ICES, 2000).

Biomass - The biomass estimate for piscivorous seabirds of 0.25 $t \cdot km^{-2}$ is a reduction of the Chesapeake Bay model estimate (Sellner *et al.*, 2001).

Diet compositions - The diet composition for piscivorous seabirds was taken from the Chesapeake Bay model and was modified by reducing predation on menhaden and increasing imports based on the large number of migratory seabirds.

Non-Piscivorous seabirds

Q/B - The consumption ratio estimate of 120 $year^{-1}$ was from data for the non-piscivorous seabirds group in Preikshot (2007).

P/B - A total mortality estimate for non-piscivorous seabirds of 0.51 $year^{-1}$ was taken from the Chesapeake model and was based on annual mortality rate of 37% for mallard males and 44% females (Anderson, 1975).

Biomass - The biomass estimate for non-piscivorous seabirds of 0.121 $t \cdot km^{-2}$ was taken from the Chesapeake Bay model and was based on advice provided in a Chesapeake Ecopath Workshop (Sellner *et al.*, 2001).

Diet compositions - The diet composition for non-piscivorous seabirds was taken from the Chesapeake Bay model.

INVERTEBRATES

Blue crabs

Q/B- The consumption ratio of 4.0 $year^{-1}$was taken from the Chesapeake Bay model.

P/B – The Delaware Bay model utilized a P/B= 1.21 $year^{-1}$. This was based on a stock assessment for Delaware Bay that used a natural morality of M = 0.8 $year^{-1}$ assuming a lifespan of 4 years (Kahn, 2003) and fishing mortality on total stock (recruits and post recruits) was F = 0.41 $year^{-1}$ (2000-2002).

Biomass – Biomass was estimated using an ecotrophic efficiency of 0.95.

Diet – Diet taken from Chesapeake Bay model, averaged across stanzas.

Hard Clams

Q/B - The consumption ratio was estimated to be 5.1 $year^{-1}$ assuming a P/Q = 0.20 (Chesapeak Bay Model)

P/B - A total production/biomass ratio of 1.681 $year^{-1}$ was calculated using Brey's Multi-parameter P/B model (Brey). This assumes an average mass of 20 g, water T = 15 °C, non-motile behavior, an average water depth of 1.5 m, and a joules to biomass conversion ratio of 1.28J per mg of wet weight with shell (Brey et al 2010, see conversion worksheet).

Biomass – 26.18 t/km2. This is based on a density of 1,309,233 clams per km^2 (adjusted values for the 1985-1987 surveys, Celestino 2002) and an average mass of 20 g (mean length of 7.46cm, Celestino 2013, length to weight average relationship verified 10/27/13 by JV in supermarket).

Diet – Diet taken from Chesapeake Bay model.

Oyster

Q/B - The Q/B ratio of 2.0 year^{-1} was taken from the adult stanza of the Chesapeake Bay Model.

P/B – A 2009 survey of the restored oyster reef at Good Luck Point determined a mean annual mortality of 47%, or an M=0.63 year^{-1} (Calvo 2010). As oysters in Barnegat Bay are an unfished resource, Z=M=.63 year^{-1}.

Biomass – Based on NJDEP experience there does not appear to be a viable oyster set in Barnegat Bay; the known oyster reef is seeded by the NJDEP. In order to keep oysters in the model for future management considerations the biomass was set to 0.001t/km^2 to simulate a very small population.

Diet – Data taken from the Chesapeake model.

Sea Nettles

Q/B – A Q/B of 20 year^{-1} was taken from the Chesapeake Bay model. This value is based on an assumed P/Q of 0.25.

P/B – As reported in the Christensen et al (2006), Matishov and Denisov (1999) estimated a daily growth rate for *Aurelia aurita* of 0.053 at 5 ºC to 0.15 at 16.5 ºC. Sea nettle medusa are present in the Barnegat Bay during the summer months, when waters are typically warmer than 16.5 ºC. As such the P/B for Barnegat Bay was calculated as (0.15*365)/4 ~ 13 year^{-1}.

Biomass – A biomass of 1.38 t/km^2 (0.92 under old volume) was calculated using bay-wide survey data from Monmouth University for 2012 and an average wet weight of 56g for individuals between 35mm-144mm. Because there are no reports of sea nettles in Barnegat Bay until the later 1980s -early 1990s this initial population is completely removed via “dummy” fishing fleet, whose effort is reduced over time.

Diet – The sea nettle diet data was taken from the Chesapeake Bay model (no citations given)

Ctenophores

Q/B - Shushkina *et al.* (1989) found that ctenophores in their study had growth rates 1.5 to 2 times greater than true jellyfish. Therefore, the Q/B value for ctenophores was the value for sea nettles multiplied by 1.75, *i.e.*Q/B was 35 year^{-1}.

P/B – Shushkina *et al.* (1989) found that ctenophores in their study had growth rates 1.5 to 2 times greater than true jellyfish. Ctenophores tend to be present in Barnegat Bay at cooler temperatures than those of sea nettles, therefore the P/B was calculated as 1.75 times the average estimated daily growth rate of *Aurelia aurita* over the course of 3 months ((((0.053+0.15)/2)*365)/4)*1.75 ~ 16.2 year^{-1}.

Biomass – A biomass of 7.86 t/km^2 was calculated using bay-wide survey data collected by Monmouth University during 2012 and an average weight of 3.42g per individual.

Diet - The ctenophore diet data was taken from the Chesapeake Bay model (no citations given)

Benthic infauna/epifauna (shrimp, worms, non-blue claw crabs)

Q/B – A consumption ration of 5.0 $year^{-1}$ was estimated by Ecopath after designating a P/Q ratio of 0.2, as taken from the Chesapeake Bay Model.
P/B – A P/B of 2.0 $year^{-1}$ was taken from the Chesapeake Bay model.
Biomass – Estimated by Ecopath, based on a group ecotrophic efficiency of 0.9 as taken from the Chesapeake Bay model.
Diet – Diet data taken from Chesapeake Bay model.

Amphipods

Q/B – Ecopath estimated a Q/B = 5.0 $year^{-1}$ using a P/Q ratio of 0.2, following the Chesapeake Bay model.
P/B – A P/B of 3.8 $year^{-1}$ was used based on the average P/B of *Ampelisca abdita* at 3 locations within Jamaica Bay (Franz and Tanacredi 1992). *A. abdita* was the most common amphipod found in Barnegat Bay sampling in 2012.
Biomass – The biomass of amphipods was estimated by Ecopath using an EE=0.900. We attempted to utilize the first year of NJDEP Barnegat Bay research program data, which is the only study of amphipod density bay-wide, though it is restricted to summer sampling only. A 1974/1975 study (Haskin and Ray 1979) documented amphipod density throughout the year, but on a limited spatial scale. In the 1974/75 study the average yearly density across all sites was approximately 2.5 times larger than the summer density during the same time period. To estimate amphipod biomass, the average density of the 2012 study was multiplied by 2.5, and the resulting density multiplied by the weight of an average amphipod (0.003g) to reach an estimate of 1.53g/m^2. This empirically determined biomass is approximately one-half of the biomass required to balance the model as found by Ecopath.
Diet – The diet data for this group was taken from the benthic infauna group.

Copepods (Mesozooplankton)

Q/B – A consumption ration of 83.333 $year^{-1}$ was estimated by Ecopath after designating a P/Q ratio of 0.3, as taken from the Chesapeake Bay Model.
P/B – A mortality rate of 25 $year^{-1}$ was taken from the Chesapeake Model, as estimated during the Chesapeake Bay Ecopath Workshop (1989).
Biomass – Copepod biomass was estimated using an ecotrophic efficiency of 0.95.
Diet – The diet ratio, 72% microzooplankton, 28% phytoplankton is from the Chesapeake Bay model.

Microzooplankton

Q/B – A consumption ration of 350 $year^{-1}$ was estimated by Ecopath after designating a P/Q ration of 0.4, as taken from the Chesapeake Bay Model.
P/B – A total mortality rate for microzooplankton of 140 $year^{-1}$ was taken from the Chesapeake Bay model.
Biomass – Biomass was estimated based on an assumed EE of 0.95.
Diet – The 100% phytoplankton diet follows the Chesapeake Bay model.

Phytoplankton

P/B – We elected to use the Chesapeake value of 160 $year^{-1}$ over the Delaware Bay value of 60 $year^{-1}$ as the Chesapeake is a highly eutrophic system more similar to the conditions found in Barnegat Bay.

Biomass – An estimated wet weight of 7.705 t/km2 was calculated using the August 2011 to September 2012 data (ugC/L) collected as part of the Governor's Barnegat bay Initiative and a conversion ratio of 10 mg wet weight:mg C (Emax report, Dalsgaard and Pauly 1997). However, this biomass is far too small to support the grazing pressure calculated. The minimum biomass required to balance the model assuming an ecotrophic efficiency of 0.95 is 25.2 t/km^2, which is in-line with the estimates for the Chesapeake Bay.

Benthic algae

P/B – The Chesapeake model assumed a value of 80 $year^{-1}$.

Biomass – Biomass of benthic algae was estimated based on an assumed EE of 0.9 (Chesapeake).

SAV

P/B – Mortality for *Z. marina* was estimated in the Chesapeake as Z = P/B =5.11 $year^{-1}$, which was taken from a similar system in Japan (Oshima *et al.*, 1999).

Biomass – In 1979 there was approximately 8,053 ha of mapped submerged aquatic vegetation (Northern segment: 767, Central segment: 5,126, Southern segment: 2,160) out of the 27,900 hectares of Barnegat Bay (Lathrop et al 2001). The highest recorded annual eelgrass maximum biomass in the southern and central portions of the bay occurred in 2004 and was 219.7 g dry wt /m^2, while the highest Ruppia biomass recorded in the northern segment occurred in 2011 and was 32.8 g dry wt/ m^2 (Kennish et al 2013). Expanding the biomass estimates over the 1979 SAV acreage yields a baywide total biomass of 1,625.891t, or 5.82t/km^2

Appendix 2 – Ecopath Initial Diet Composition

	Piscivorous seabirds	Non-piscivorous	weakfish	striped bass	summer flounder	bluefish	winter flounder	Atlantic silversides	Atlantic croaker	spot	Atlantic menhaden
Piscivorous seabirds											
Non-piscivorous											
weakfish	0.0056			0.2		0.013					
striped bass	0.0166										
summer flounder	0.011										
bluefish	0.02										
winter flounder	0.0058				0.2						
Atlantic silversides	0.017		0.05	0.221	0.132	0.087					
Atlantic croaker			0.005	0.01		0.005					
spot			0.03			0.011					
Atlantic menhaden	0.1			0.206		0.255					
river herring	0.028										
mummichog	0.03					0.36					
bay anchovy	0.07		0.535	0.2	0.273	0.094	0.018				
benthic infauna/epifauna		0.276	0.352	0.06	0.186	0.066	0.742	0.59	0.8	0.509	0.18
amphipods			0.022				0.07	0.244		0.25	
blue crabs	0.004		0.006	0.1	0.2	0.103	0.002				
hard clams		0.01		0.003			0.157			0.057	
oysters											
copepods								0.154	0.2	0.18	0.338
Microzooplankton											
sea nettles											
ctenophores											
benthic algae											
phytoplankton											0.421
SAV		0.128									
detritus		0.011			0.009	0.006	0.011	0.012		0.004	0.061
import	0.692	0.575									

	river herring	mummichog	bay anchovy	benthic infauna	amphipods	blue crabs	hard clams	oysters	copepods	Micro zoo	sea nettles	ctenophores
Piscivorous seabirds												
Non-piscivorous												
weakfish												
striped bass												
summer flounder												
bluefish												
winter flounder												
Atlantic silversides												
Atlantic croaker												
spot												
Atlantic menhaden												
river herring												
mummichog												
bay anchovy											0.054	
benthic infauna/epifauna	0.435	0.260		0.02	0.02	0.5						
amphipods	0.055	0.170	0.044									
blue crabs						0.125						
hard clams						0.175						
oysters												
copepods	0.5	0.19	0.582								0.421	0.666
Microzooplankton			.370	0.08	0.08				0.72			0.334
sea nettles												
ctenophores											0.525	
benthic algae		0.12		0.3	0.3	0.05	0.5					
phytoplankton	0.005			0.4	0.4		0.25	0.99	0.28	1		
SAV						0.05						
detritus	0.005	0.26	0.004	0.2	0.2	0.1	0.25	0.01				
import												

Appendix 3 - Landing Calculations for the Barnegat Bay Ecopath Model

Directed Fisheries

The National Marine Fisheries Service (NMFS) commercial landings database is the most comprehensive record of commercial landings available for the time period of interest (1950-2011). However, these data represent landings for all of New Jersey, and are not Barnegat Bay specific. The NMFS landings data used below are a subset of the statewide landings based on gear that could be used within an estuary. Gear types considered usable in the bay include the following: by hand; cast nets; dip nets, common; fyke and hoop nets, fish; hand lines, other; pots and traps, blue crab; and weirs. Because these gear types have been used in the Barnegat Bay as well as other larger estuaries throughout the state (Raritan Bay, Delaware Bay, *etc.*), this subset likely overestimates commercial removals from Barnegat Bay. Where Barnegat Bay specific landings data are available they were used to the maximum extent possible.

Recreational landings for finfish were taken from the NMFS Marine Recreational Fisheries Statistics Survey (MRFSS) for Ocean County, inland waters only. The landings for 1981 were used to initialize the model as that is the earliest year for which data is available.

The source and calculations for each species are described below.

Atlantic croaker – Based on the subset of NMFS commercial landing data, there was no harvest of Atlantic croaker reported in the 1980s. There were no recreational landings of croaker reported for Ocean County.

Atlantic Menhaden - There was no commercial harvest of menhaden recorded in the NMFS landing data for the gear types used in Barnegat Bay in 1980. There were no recreational landings of menhaden reported for Ocean County in the MRFSS database. Menhaden are commonly used as bait in the recreational fishery in Barnegat Bay, therefore an estimated landing of 0.2MT was attributed to the recreational fishery, though this likely underestimates landings.

Blue Crab – In Barnegat Bay the commercial blue crab fishery can be divided into a winter dredge fishery and a pot/trap line fishery in the remainder of the year. Landings data specific to Barnegat Bay were available from the NJDEP for 1995-2011, while statewide landings were available from NMFSS for 1980-2011. The NJDEP data was regressed on the NMFS data and the results used to calculate bay specific total landings for 1981-1994. The winter dredge fishery represented approximately 17% of the baywide total (NJDEP data); this ratio was used to estimate the gear specific landings from the total baywide landings of 221 metric tons for 1981. Therefore the winter dredge fishery in 1981 landed an estimated 38.1 metric tons while the pots and trot lines accounted for an estimated 183.3 metric tons. In 2007 the recreational harvest of blue crabs in Barnegat Bay was estimated to be 80% of the total commercial harvest (B. Muffley personal communication), leading to an estimated recreational harvest of 177.1 metric tons in 1981.

Bluefish – Barnegat Bay specific commercial landings were available for bluefish for 1997 only (Kennish SCR). The bay specific landings represented 21% of the subset landings for that year (NMFS). That ratio was utilized to calculate an estimated Barnegat Bay specific commercial landing of 0.02 metric tons for 1980. In 1981approximately 209.1 metric tons of bluefish were landed in Ocean County inland waters (MRFSS).

Hard Clam – Hard clams are historically one of the most important commercial fishery resources in Barnegat Bay. Hard clam landings from Barnegat Bay approached 226.8 metric tons in 1980, the closest year for which data was available (G. Calvo, personal communication of NMFS data, 2011). There are no estimates of hard clam recreational landings available.

River herring – Alewife and blueback herring have been combined into this single category given the similarities in their life history strategies and propensity to co-migrate. In 1981 there were no commercial landings of either species in the subset landings, and no landings reported for Ocean County's recreational inland fishery. However, there were known fisheries for river herring within the bay associated with bait collection. As such a total landing of 0.1MT was assumed based on the landings in subsequent years and split evenly between the recreational and commercial sectors.

Spot –There were no commercial landings of spot recorded in the subset landing data for the late 1970s through mid 1980s. There were 1.1 metric tons of spot landed in the Ocean County inland recreational fishery in 1981.

Striped Bass – In 1981 there were no commercial landings of striped bass recorded in the subset landing data. There were no landings reported for Ocean County's recreational inland fishery. However, there was a well-documented recreational fishery present at the time, therefore 26 MT was used , which is the average of reported landings from 1981-201.

Summer flounder – Commercial landings of summer flounder approached 0.2 metric tons in 1981 according to the subset NMFS database. There were 224.4 metric tons of summer flounder landed in the Ocean County inland recreational fishery in 1981.

Weakfish - Barnegat Bay specific commercial landings were available for weakfish for 1993 only (Kennish SCR). The bay specific landings represented approximately 5.2% of the gear specific statewide landings for that year (NMFS landing data). That ratio was utilized to calculate an estimated Barnegat Bay specific commercial landing of 0.078 metric tons for 1981. There were 3.29 metric tons of weakfish landings reported for Ocean County's recreational inland fishery in 1981.

Winter flounder – The NJDEP Bureau of Marine Fisheries estimates a commercial harvest of approximately 10.68 metric tons of winter flounder from Barnegat Bay in 1981. In 1981 there were 247 metric tons of winter flounder landed in the Ocean County inland recreational fishery.

OCNGS

The Oyster Creek Nuclear Generating Station "landings" info can be divided into two categories, impingement/impingeable size losses and entrainment losses. Impingement losses describe those animals that become trapped on the traveling Ristroph screens (9mm mesh) associated with the Circulating Water Intake

Structure (CWIS) and are subsequently deposited into a fish return system and into the discharge canal. Impingeable size losses are biota that are large enough to be impinged on the Ristroph screens if they were present at the Dillution Water Intake Structure (DWIS). Entrainment losses are the biota that pass through the CWIS and DWIS structures and pass through the plant and dilution pumps, respectively. The data used to estimate these values were collected as part of periodic relicensing of the facility, and were most recently collected during 2005-2007 and include in the "Characterization of the aquatic resources and impingement and entrainment at Oyster Creek Nuclear Generating Station" September 2008.

Impingement/Impingeable size losses

During 2006-2007 the estimated annual biomass of the young of year (YOY) and older ages of selected fish and crustaceans impinged on the traveling screens at the CWIS was calculated (Appendix A: Detailed Characterization of the aquatic resources and impingement and entrainment at Oyster Creek Nuclear Generating Station, Tables A-7 and A-8). The biomass of each species was then multiplied by the empirically determined impingement mortality rate (Appendix H, Tables H-2 and H-4) to derive a CWIS impingement mortality (kg/yr). The estimated annual biomass of impingeable sized fish and shellfish that were entrained through the DWIS was calculated (Tables A-15 and A-18) and multiplied by the empirically determined mortality rates (Tables H-5 and H-6) to derive a DWIS impingeable size mortality (kg/yr). It should be pointed out that the mortality rates were instantaneous, that is injured individuals were considered "live" at the time of counting, and thus the mortality rates are likely low.

Entrainment losses

Entrainment losses occur when biota are able to avoid or slip through the traveling screens at the CWIS and are carried through the cooling water system or are taken up by the DWIS. The number of individual fish in each species entrained into either the CWIS (Table A-10) or DWIS (A-20) are broken into 5 size categories; eggs, yolk sac larvae, post-yolk sac larvae, YOY, and YOY+. Blue crabs were divided into adult, juvenile, and megalops (tables A-12 and A-22). For this model the entrainment analysis was limited to post-yolk sac larvae, YOY, and YOY+ fish and megalops stage of blue crab. Biomass for each species/size class was calculated by taking the median or mode length from the CWIS entrainment sampling length frequency histograms (Appendix C: Impingement and entrainment studies at Oyster Creek Generating Station 2005-2007) and searching the literature for the corresponding weight. This weight was multiplied by the annual estimated number of individuals to derive an estimate of annual biomass. The biomass estimate was then multiplied by the appropriate empirically determined mortality rate to derive an estimate of entrainment losses for both the CWIS and DWIS. The latent mortality was calculated as the number of live, healthy entrainable-size specimens collected from the discharges who survived for 24 hours (Appendix F, Sections 2 and 3). The mortality was applied equally across all size classes. Given that this methodology does not take into account individuals that do not survive passage through the system it likely underestimates mortality. The specific values selected for the length, weight, and mortality rate for each species are detailed below.

Adult and juvenile blue crabs were not included in the entrainment analysis as there are a number of discrepancies in the crab data. The CWIS impingement sampling collected crabs in the 8-166mm size range; these specimens should not be able to pass through the Ristroph screen, thus nearly eliminating any entrainment at the CWIS. Further, any crabs of this size should be considered part of the "entrainment of impingeable sizes" DWIS calculations, and to include them in DWIS entrainment would be double counting.

Atlantic croaker –

Post-yolk sac – Lengths ranged from 4-16mm, with a rather uniform distribution between 7-15mm. The ASMFC 2005 stock assessment for larval croaker suggests a mode of 11mm and a weight range of 0.02 – 0.04g. An average weight of 0.03g was used in the analysis.

YOY – The lengths of YOY croaker ranged from 15-72mm, with the distribution skewed heavily to the left. The modal length was 21mm. An average weight of 0.06 grams at 21mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for croaker. The empirically determined weakfish mortality rate (CWIS 0.8, DWIS 0.75) was used as they are both Sciaenids and share similar characteristics at the larval stage.

Atlantic Menhaden

Post-yolk sac – Lengths were bimodally distributed from 6 – 33 mm, with the larger mode at 24 mm. Hettler (1976) found an average weight of 0.195 grams at 28mm.

YOY – Lengths were evenly distributed between 27-42mm , with a mean length of 34. Hettler (1976) found an average weight of 0.494 grams at 34mm.

Mortality – A 24 hour mortality rate of 1 was used for the CWIS and 0.72 for the DWIS.

Atlantic silverside -

Post-yolk sac – Lengths were unimodally distributed from 4 – 8 mm, with the mode at 5mm.

YOY – Lengths were evenly distributed between 71-85mm. The silverside should be fully recruited to the Ristroph screen at 72mm, so 71mm was selected. An average weight of 0.2.25 grams at 71mm was calculated using the length-weight regression from FishBase.

YOY+ - Lengths were evenly distributed between 74-102mm, with a mean at 87mm. An average weight of 4.71 grams at 87mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for silverside. The empirically determined bay anchovy mortality rate (CWIS 0.97, DWIS 0.94) was used as they have similar body shapes and tolerances at the larval stage.

Bay anchovy -

Post-yolk sac – Lengths were unimodally distributed from 3 – 37 mm, with the mode at 8mm. Using the length-weight relationship in Table 5 of Leak and Houde (1987), an 8mm individual is approximately 11 days old, and would have a dry weight of 0.000114g. If larvae are assumed to be 95% water, this would lead to a wet weight of 0.0023

YOY – Lengths were unimodally distributed between 26-69mm , with a modal length of 34. An average weight of 0.32 grams at 34mm was calculated using the length-weight regression from FishBase.

Mortality - A 24 hour mortality rate of 0.97 was used for the CWIS and 0.94 for the DWIS.

Summer flounder –

Post-yolk sac – Lengths were unimodally distributed from 10 – 17 mm, with the mode at 14mm. An average weight of 0.04 grams at 14mm was calculated using the length-weight regression from FishBase.

YOY – Lengths were unimodally distributed between 12-17mm , with a modal length of 14. Given the overlap in lengths with post-yolk sac, it appears the demarcation between classes is based on eye migration. An average weight of 0.04 grams at 14mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for summer flounder. The empirically determined winter flounder mortality rate (CWIS 0.88, DWIS 0.90) was used as they have similar body shapes and tolerances at the larval stage.

Weakfish –

Post-yolk sac – Lengths were unimodally distributed from 2 – 14 mm, with the mode at 5mm. Using the empirically measured mean dry weight of 0.000171g for 5mm larvae from Duffy and Epifanio (1994) leads to a wet weight of 0.0034 grams assuming 95% water.

YOY – Lengths were evenly distributed between 11-123mm , with a mean length of 36. An average weight of 0.41 grams at 36mm was calculated using the length-weight regression from FishBase.

YOY+ - The only size captured in sampling was 172mm. An average weight of 0.44 grams at 172mm was calculated using the length-weight regression from FishBase.

Mortality - A 24 hour mortality rate of 0.80 was used for the CWIS and 0.75 for the DWIS.

Winter flounder –

Post-yolk sac – Lengths ranged from 2-11mm, with a relatively uniform distribution between 3-6mm. The average length was 5mm. . Based on mean larval lengths in Buckley et al. (1991), a 6mm winter flounder is approximately 4 weeks old. Laurence (1975) determined the mean dry weight of a 4 week old winter flounder kept at a similar temperature to be 0.000206g. This leads to a wet weight of 0.00412 grams assuming 95% water.

YOY – Lengths ranged between 6-7mm, with 6mm fish dominating the catch. Given the overlap in lengths with post-yolk sac, it appears the demarcation between classes is based on metamorphosis. Laurence (1975) determined the mean dry weight of a metamorphosed winter flounder to be 0.001243g. This leads to a wet weight of 0.02486 grams assuming 95% water.

Mortality - A 24 hour mortality rate of 0.88 was used for the CWIS and .90 for the DWIS.

Blue Crab –

Megalops – There was no information provided in the OCNGS reports on the length, weight, or mortality of blue crab megalopae with regard to entrainment sampling. Blue crab instar #1 have an average carapace width of 2.5mm, which is sufficiently small enough to pass through the Ristroph screen, and have an estimated average of weight of 0.0033 grams (Newcombe et al., 1949). Mortality was assumed to be similar to that found empirically for *Mysidopsis bigelowi* during the study period of 0.66 and 0.17 for the CWIS and DWIS respectively.

Fuzzy cognitive mapping in support of integrated ecosystem assessments: developing a shared conceptual model among stakeholders.

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Abstract
Ecosystem-based approaches, including integrated ecosystem assessments, are a popular methodology being used to holistically address management issues in social-ecological systems worldwide. In this study we utilized fuzzy logic cognitive mapping to develop conceptual models of a complex estuarine system among four stakeholder groups. The average number of categories in an individual map was not significantly different among groups, and there were no significant differences between the groups in the average complexity or density indices of the individual maps. When ordered by their complexity scores, eight categories contributed to the top four rankings of the stakeholder groups, with six of the categories shared by at least half of the groups. While non-metric multidimensional scaling (nMDS) analysis displayed a high degree of overlap between the individual models across groups, there was also diversity within each stakeholder group. These findings suggest that while all of the stakeholders interviewed perceive the subject ecosystem as a complex series of social and ecological interconnections, there are a core set of components that are present in most of the groups' models that are crucial in managing the system towards some desired outcome. However, the variability in the connections between these core components and the rest of the categories influences the exact nature of these outcomes. Understanding the reasons behind these differences will be critical to developing a shared conceptual model that will be acceptable to all stakeholder groups and can serve as the basis for an integrated ecosystem assessment.

Keywords: ecosystem based management, Barnegat Bay, fuzzy logic cognitive mapping, FCM,

1.0 Introduction

It is widely accepted that the sustainable management of natural resources must include consideration of human interactions with the environment, not only from a unidirectional perspective (humans impacting natural systems or vice-versa), but with the understanding that these coupled socio-ecological systems are dynamic and have a variety of two-way interactions and feedbacks (An and Lopez-Carr 2012, Liu *et al.* 2007). The realization that the use of natural resources is inextricably interwoven with the social, political, and economic complexities of human systems has led to these management challenges being called "wicked problems" (Xiang 2013), *i.e.* "problems which are ill-formulated, where the available information is confusing, where there are many clients and decision makers with conflicting values, and where the ramifications in the whole system are thoroughly confusing" (Churchman 1967). With an ever increasing number of wicked problems recognized in social-ecological systems throughout the globe (Sayer *et al.* 2013, Jentoft and Chuenpagdee 2009, Ludwig 2001) the idea of ecosystem-based management has gained traction, particularly in marine policy in the United States (NOAA 2006). Ecosystem-based management (EBM) attempts to look at a defined geographic area in a holistic manner, defining management strategies for an entire system rather than individual components (Levin *et al.* 2009).

To successfully manage resources from an ecosystem-wide perspective it is necessary to gather pertinent information on all of the system components, but by definition the data available in instances of wicked problems are confusing, as no clear patterns are readily emergent, or if there are patterns they are often contradictory. One organizing framework to synthesize and analyze large amounts of confusing data to support EBM is the Integrated Ecosystem Assessment, or IEA (Levin *et al.* 2009). The IEA approach is a series of formal processes during which relevant stakeholder groups (including public representatives, scientists, managers and policy makers) synthesize existing knowledge regarding the ecosystem in question, set ecosystem management objectives, select management options, and then adjust future management actions based on feedback from continuing monitoring. The initial activity in the IEA process is the scoping step, during which stakeholder groups define the ecosystem to be addressed, review existing information, construct a conceptual ecological model that identifies ecosystem attributes of concern and relevant stressors, and develop appropriate management objectives (Levin *et al.* 2008). Generally, this step is conducted during one or more workshops (Hobbs *et al.* 2002, McClure and Ruckelshaus 2007) where participants interact in a facilitated format designed to generate consensus on the ecosystem attributes and management objectives. However, there are concerns with the quality of both the process and the outcome when public participation is included in solving environmental issues (NRC 2008). In particular, prior studies have shown that groups tend to converge on majority views, that powerful or influential individuals or groups may attempt to dominate or unduly influence the proceedings, and that quality processes and outcomes, especially those related to consensus building, can be cost prohibitive (NRC 2008).

In light of the potential problems described above, there is a clear need for a strategy that can combine traditional scientific knowledge with public local context, thereby reducing uncertainty and providing for a diversified and adaptable knowledge base (Raymond *et al.* 2010, Gray *et al.* 2012). One methodology that has been suggested is Fuzzy Logic Cognitive Maps (FCMs) (Axelrod 1976). FCM are a simplified way of mathematically modeling a complex system (Özesmi and Özesmi 2004), and have been used to represent both individual and group knowledge (Gray *et al.* 2012). This approach has been applied to processes and decisions in human social systems, the operation of electronic networks, and in the ecological realm to identify the interactions between social systems, biotic, and abiotic factors in lakes (Özesmi 2003, Hobbs *et al.* 2002), coal mine environs (Zhang *et al.* 2013), farming systems (Vanwindekens *et al.* 2013), nearshore coastal zones (Meliadou *et al.* 2012, Kontogianni *et al.* 2012) and the summer flounder fishery (Gray *et al.* 2012), but applications in estuaries has been rare.

In this paper we investigate if fuzzy logic cognitive mapping can be used to develop a shared conceptual model among various stakeholder groups that can serve as the basis for an integrated ecosystem assessment in a complex estuarine system. We first develop conceptual ecosystem models for different stakeholder groups using FCM. Next we combine those models into a shared conceptual ecosystem model. A shared understanding of the important components and processes of the ecosystem in question is critical if stakeholder groups are to fully "buy-in" to future management decisions (Ogden *et al.* 2005). The FCM methodology ameliorates many of the challenges associated with integrating the different types of stakeholder knowledge (Gray *et al.* 2012), and the transparent nature of the model combination allows stakeholders to identify how each groups' model contributes to the overall understanding. We do not expect the different groups' conceptual models to share all of the components; rather we anticipate these differences to be highly informative. Indeed, understanding why these differences occur is likely to help us avoid misunderstandings and disagreements during future phases of the IEA process (Kontogianni *et al.* 2012b). Therefore, we analyze the components and structural similarities and differences among the models to assess the utility of this approach as the basis for the IEA scoping process, with the understanding that the scoping process is an essential first step toward effective EBM.

2. Methodology

2.1 Study Site

The social ecological system we have chosen to study is the Barnegat Bay, a 279 km^2 lagoonal estuary located in central New Jersey, USA (Figure 1). The surrounding 1,730 km^2 watershed is home to an estimated 580,000 year round residents (US Census Bureau 2012), with a summer population that swells to over 1 million with the influx of tourists. The physical setting of the watershed is well described by Kennish (2001), but points germane to our study are repeated here. Land use is a mix of urban and suburban uses in the northeast and along the barrier islands, grading to less sparsely populated forested areas to the south and west. Portions of the E.B. Forsythe National Wildlife Refuge and the Pinelands National Reserve are located along the eastern and western sides of the watershed, respectively. There is limited extractive and agricultural land use, and other than minor hard clam and blue crab fisheries, no real commercial fishing. The watershed is considered "highly eutrophic" (Bricker *et al.* 2007), mainly due to nutrient enrichment through non-point source pollution, and the nation's oldest continuously operating nuclear power plant, Oyster Creek Nuclear Generating Station, is located within the watershed. There is extensive recreational use of the bay's waters for fishing, boating, sailing, and to a lesser degree, bathing.

2.2 Data collection

FCMs are models of a how a system operates based on key components and their causal relationships. The components can be tangible aspects of the environment (a biotic feature such as fish or an abiotic factor such as salinity) or an abstract concept such as aesthetic value. The individual participants identify the components of the system that are important to them, and then link them with weighted, directional arrows. The weighting can range from -1 to +1 (Hobbs *et al.* 2002, Özesmi and Özesmi 2004, Gray *et al.* 2012), and represents the amount of influence (positive or negative), that one component has on another.

To collect FCM from a wide variety of stakeholders with knowledge of the Barnegat Bay ecosystem we contacted the Barnegat Bay Partnership, a US Environmental Protection Agency National Estuary Program, to obtain a list of their management and science committee members, as well as a list of public citizens who have expressed long-term interest in the ecosystem. While the map of an individual stakeholder provides information regarding that particular individual's conception of the important components and linkages within the system, it can be combined with other individuals within the group to produce a more robust picture of the group's understanding of the system (Özesmi and Özesmi 2004). In addition, all of the individual stakeholder maps can be combined into a single map depicting the collective understanding of the system. To this end, the individuals were divided into four groups that were determined *a priori*: scientists (n=19), managers (n=11), environmental non-governmental organizations (n=6), and local residents (n=6) (Table 1). These groups were selected to represent several (though not all) of the major categories of stakeholders present in ongoing efforts to manage and improve the bay's natural resources. The scientist group consisted of individuals from academia, state, and federal institutions who have conducted research within the Barnegat Bay watershed, while managers were from federal, state, county, or local natural resource management agencies who had jurisdiction on some form of activity within the watershed. Environmental non-governmental organizations included local, statewide, and regional groups who are active in watershed protection. The local residents were referred to us by other interviewees, and included commercial and recreational fisherman as well as private citizens with a long-standing interest in the bay.

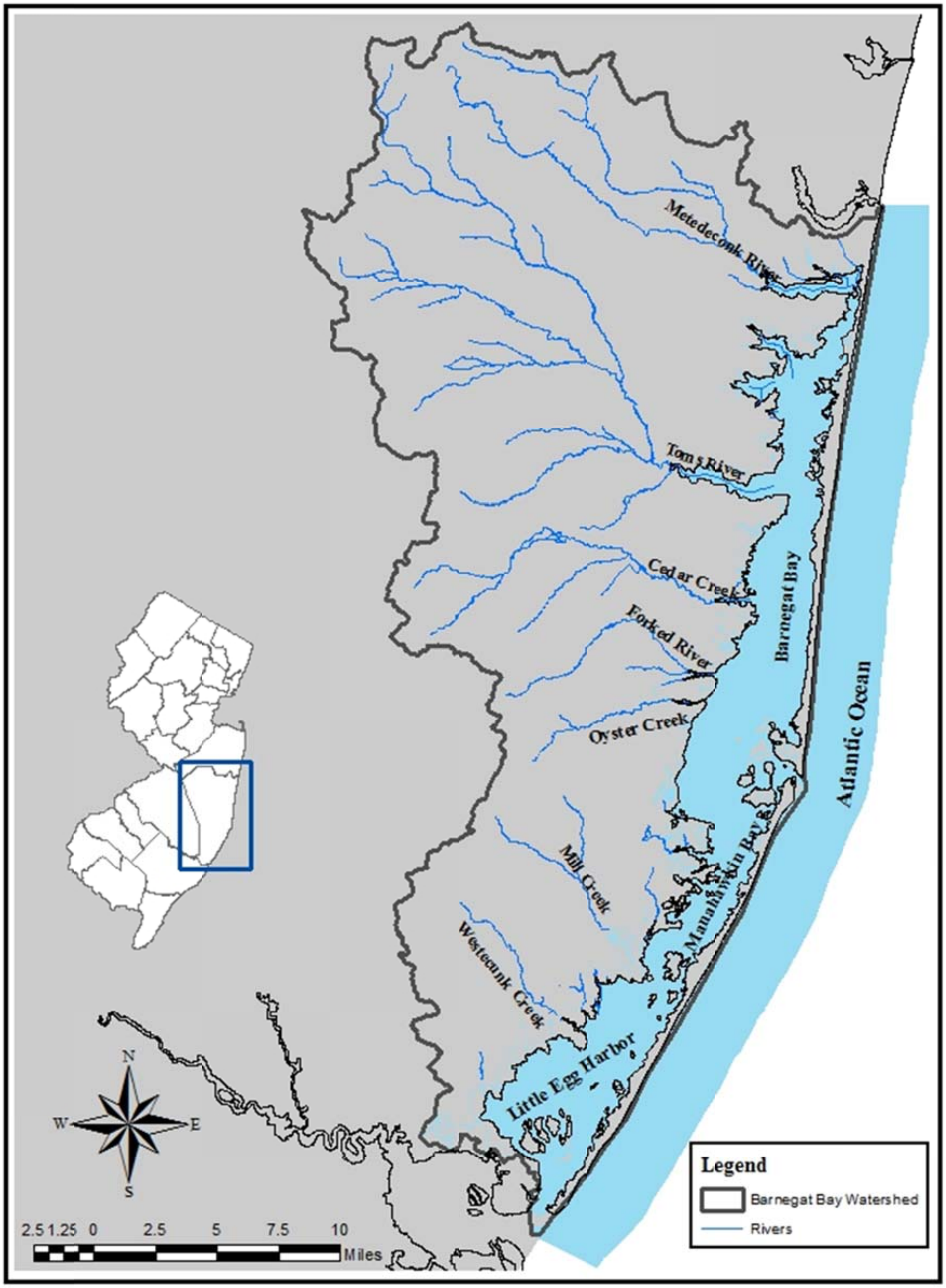
Table 1: Information on stakeholders who completed fuzzy cognitive maps on the Barnegat Bay social-ecological system			
Stakeholder group	Maps (N)	People (N)	Occupation/organization/social group
Scientists	19	19	Academic scientists, federal and state agency research scientist
Managers	11	11	Federal, state, county, and local resource managers
Environmental NGOs	6	6	Regional, statewide, and local environmental non-profits
Local people	6	6	Baymen, commercial fisherman, recreational fisherman, longtime (+40 year) residents

In accordance with the procedures used in prior studies (Carley and Palmquist 1992, Özesmi and Özesmi 2004, Gray *et al.* 2012) individuals were interviewed separately, and each interview began with an overview of the project, a promise of anonymity, and an example of a simple FCM related to an issue outside of the realm of ecology, namely traffic flow. Interviewees were then asked to describe what they considered to be the key components of the Barnegat Bay social-ecological system and how those components relate to one another. They were then asked to score the strength and direction of the relationship using positive or negative; high, medium, or low. The discussion continued until the interviewee was satisfied that the map as drawn accurately depicted their understanding of the system. This ranged anywhere from 45 minutes to 180 minutes, with the typical session lasting 90 minutes. Once mapping was complete, the interviewees were asked which of the components in their maps they would like to see increased and which decreased. The interviews were conducted under an approved human subjects protocol (number: E13-560).

2.3 Data Analysis

There are a number of different methods that can be used to analyze the data contained within an FCM, many of which are based upon graph theory (Harary *et al.* 1965, Özesmi and Özesmi 2004, Kosko 1991). To better understand the structure of an individual FCM we translated each map into a square adjacency matrix, with all of the variables acting as potential transmitters (influencing other variables) v_i on the vertical axis and the same set of variables acting as receivers (influenced by other variables) v_j on the horizontal axis (see Supplemental Figure 1 for an example). A list of all individual variables mentioned throughout the process was compiled and redundant variables (plurals, different names for the same species, *etc.*) were eliminated. When two variables represented opposite directions of the same concept (*i.e.* dam construction and dam removal) the more prevalent variable was retained and the other variable was renamed, with the polarity of the interactions reversed, in keeping with accepted practices (Kim and Lee, 1998). The interactions strengths between variables were then scored, with high interactions scored as 0.75, medium as 0.5, and low as 0.25 (Harary *et al.* 1965).

Figure 1 – Map of Barnegat Bay watershed with New Jersey inset.



To more easily understand the components and patterns within an individual FCM it is often helpful to simplify the map by reducing the number of variables (Harary *et al.* 1965). After all of the maps were completed we listed the full set of variables and identified those most often mentioned. We then subjectively combined less frequently mentioned variables into larger categories based on shared characteristics, a process known as qualitative aggregation. For example, "homes", "urban development", "housing", and "overdevelopment", were combined, with a number of other similar variables, into a category called "development".

With the large list of variables reduced into broader categories, the type of categories, and number of each, were identified to provide additional insight into the overall structure of the map and how these categories relate to each other (Bougon *et al.* 1977, Eden *et al.* 1992, Harary *et al.* 1965). Each category was classified as transmitter, receiver, or ordinary (both influenced by and influencing other categories), based on its indegree and/or outdegree (Table 2). Indegree is the cumulative strength of the connections entering the category (sum of the absolute values within a column in the matrix), while outdegree is the cumulative strength of the connections exiting the category (sum of the absolute values within a row in the matrix) (Özesmi and Özesmi 2004). A transmitter category has positive outdegree and no indegree, a receiver category has no outdegree and a positive indegree, and an ordinary category has positive indegrees and outdegrees (Bougon *et al.* 1977). Finally, the centrality, or a measure of a category's connectedness to other categories within the map, as well as the overall strength of those connections, was calculated as the sum of the indegree and outdegree values of a given category (Harary *et al.* 1965).

Table 2: Fuzzy Cognitive Map Indices

Term	Definition
Indegree	Cumulative strength (absolute value) of the connections entering a category
Outdegree	Cumulative strength (absolute value) of the connections exiting a category
Centrality	Sum of the indegree and outdegree for a given category
Receiver	A category with a positive indegree and no outdegree
Transmitter	A category with a on indegree and a positive outdegree
Ordinary	A category with positive indegree and outdegree
Complexity	The ratio of receiver categories to transmitter categories within a map (R/T)
Density	The number of connections within a map divided by the total connections possible between categories (C/N^2)

Indices of complexity and density were also determined for each stakeholder map. The complexity of a map is calculated as the ratio of receiver categories to transmitter categories (R/T). A large number of receiver categories in a map suggests a system where there are multiple outcomes (Eden *et al.* 1992), while a large number of transmitter categories suggest that a system is hierarchical in nature, and driven by "top down" thinking (Özesmi and Özesmi 2004). Density describes how well connected categories are within the map, and is determined by dividing the number of connections present by the maximum number of connections possible (Hage and Harary, 1983). A dense map suggests that an interviewee (or stakeholder group) perceives a number of possible pathways to influence a variable in their map (Özesmi and Özesmi 2004).

In addition to developing indices for each individual map, maps were combined 1) within stakeholder groups to produce four group maps and 2) across all individuals to produce a community map. To combine maps the connection values between two given categories are added, so connections represented in multiple maps are reinforced (provided they have similar signs) while less common connections are not reinforced, but are still included in the map (Özesmi and Özesmi 2004). To compare connection values across group maps, the summed values are divided by the number of individuals in the group.

Non-metric multidimensional scaling (nMDS) was used to assess the similarities between individual stakeholder maps (R v3.0.2). This technique orders samples by rank similarity along their two most important latent gradients and has an advantage over other ordination techniques in that it has a greater ability to accurately represent complex relations among samples in two-dimensional space (Clarke and Warwick 2001). The nMDS data were calculated as each category's centrality score for an individual stakeholder and then the Bray Curtis index was used to construct the sample similarity matrix (variable by stakeholder array). The nMDS plot was then visually assessed to identify patterns between stakeholder groupings.

Besides understanding the structure of the stakeholder groups' and community maps, maintaining the initial conditions through time allows us to determine if the model will coalesce around a stable state, go into a limit cycle, or enter into a chaotic pattern (Dickerson and Kosko 1994). To generate this steady state, the adjacency matrix of the cognitive map is multiplied by an initial steady state vector (a value of 1 for each element of the vector). The resulting vector is then subject to transformation using a logistic expression ($1/(1 + e^{-1\times x})$) to bound the results in the interval [0,1] (Kosko 1987). This new vector is then multiplied by the original adjacency matrix and again subject to the logistic function, repeating these steps until an end result is reached.

If the model reaches a steady state outcome, it is then possible to run hypothetical "what-if" scenarios to compare the function of the various models. The hypothetical scenario developed for our simulation was to maintain the category "development" at 0, which is a possible policy prescription, albeit a potentially unpopular one. To do this we utilize the process described above to determine the stable state, but this time the value of the category "development" in the vector is maintained at 0 in each time step. Setting the value of a category of interest in the multiplication vector between 0 and 1 at each time step was referred to as "clamping" by Kosko (1986). The difference between the values of the final vector of the clamped procedure compared to the steady state vector describe the relative change to the conceptual system given the framework provided by each stakeholder group. A conceptual schematics of map aggregation and steady state calculations are provided in Supplemental Figure 1 and a flow diagram of the steps in the data analysis process is provided as Supplemental Figure 2.

3.0 Results

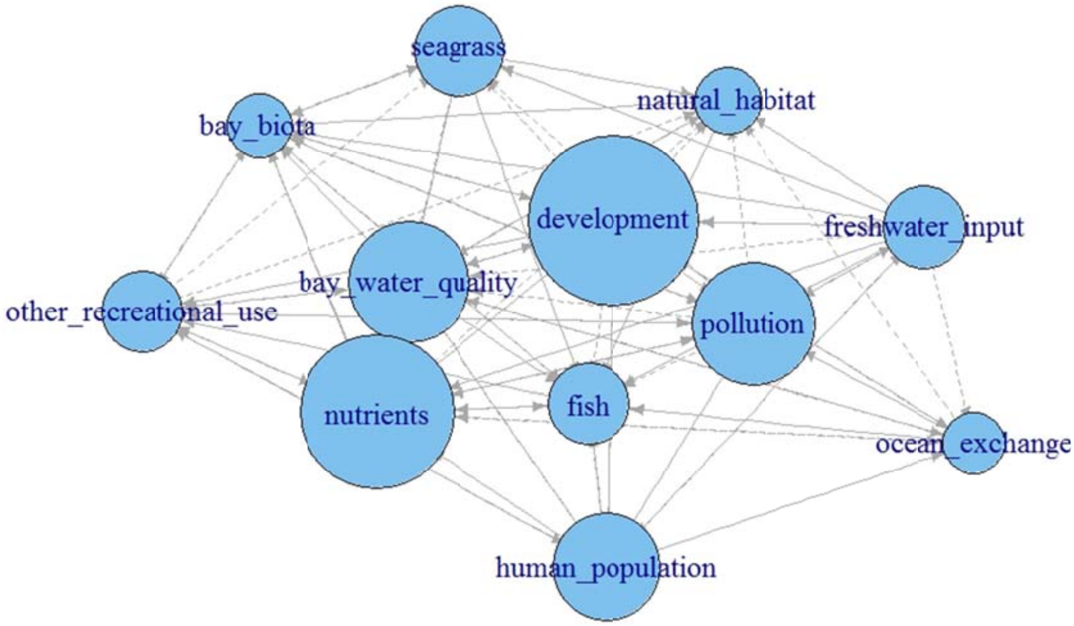
We created fuzzy cognitive maps for 42 individuals from the four targeted stakeholder groups (Table 1). The stakeholders identified 346 unique variables as important to understanding the Barnegat Bay social – ecological system, which were then aggregated into 84 categories for further analysis. Individual maps contained an average of 25 variables, which when aggregated led to an average of approximately 20 categories per map. The average number of categories in an individual map was not significantly different among groups, with the exception of NGOs ($p = 0.02$), who had an average of nearly 30 categories per map (Table 3). An examination of the accumulation curves for the total number of categories versus the number of interviews shows that the managers and scientists were well sampled, while the NGO and local residents' curves had not yet flattened out (Supplemental Figure 3). All of the NGOs active in the watershed at the time of the study were interviewed, limiting the number of samples of available. The pool of potential interviewees who met the criteria for the local resident group was also limited in size. However, the trajectories of these two groups is similar to that of the scientists and managers, suggesting that few new categories would have been added through additional interviews.

There were no significant differences between the groups in the average complexity (df=38, p=0.492) or density (df= 38, p=.129) indices of the individual maps (Table 3). The environmental NGOs and local residents had slightly higher complexity scores (more receiver categories) than the other two groups, while the managers and scientists had slightly higher average densities. The community map, by definition, contained the full suite of categories, but had an order of magnitude more connections than the group maps, leading to a map with the most interconnections between categories, and therefore the highest density. The increased number of interconnections in the community map led to all of the categories being classified as "ordinary" (i.e., both a transmitter and a receiver), with the exception of biodiversity, which was a receiver category. A subset of the community map that includes the categories with centrality scores greater than one, and their interconnections, is shown in Figure 2. For a complete list of all variables and their centrality scores please see Table S1 in the supplemental information.

Table 3: Graph indices by stakeholder group. All values, except for number of maps, are mean and standard deviation.

	Scientists	Managers	Environmental NGOs	Local people	Community
Maps	19	11	6	6	42
Number of categories (N)	20.6 (4.3)	21.2 (5.3)	29.8 (13.4)	19.3 (3.6)	84
Number of transmitter categories (*T*)	5.1 (2.7)	4.4 (2.7)	5.8 (3.3)	4.7 (2.5)	0
Number of receiver categories (*R*)	3.2 (2.8)	2.3 (1.9)	4.5 (2.9)	4.3 (1.8)	1
Number of ordinary categories	12.3 (4.3)	14.5 (4.0)	19.5 (10.8)	10.3 (2.7)	83
Number of connections (C)	38.3 (13.3)	49 (17.8)	64 (40.7)	29.5 (9.3)	1071
C/N	1.9 (0.5)	2.3 (0.6)	2.1 (0.5)	1.5 (0.4)	12.75
Complexity (*R/T*)	0.7 (0.8)	0.6 (0.5)	0.9 (0.5)	1.1 (0.6)	
Density	0.09 (0.03)	0.11 (0.04)	0.08 (0.03)	0.08 (0.02)	0.15

Figure 2: Subset of the community conceptual model. The twelve nodes with centrality scores greater than 1.0 are shown. Node size is related to centrality score, solid lines are positive interaction strengths, dotted lines are negative interactions strengths.



When ordered according to their centrality scores, eight different categories contributed to the top 4 rankings of the stakeholder groups, and six of the categories were shared by at least half of the groups (Table 4). Development had the strongest interactions for managers and local residents and was second only to nutrients for scientists and NGOs. Pollution, bay water quality, seagrass, and human population were also key shared categories, though the strength of the interactions, and their ranking, varied between groups. The outdegree strength for development and human population was at least two times that of the indegree, while pollution and bay water quality had indegrees slightly larger than outdegrees. The direction and magnitude of the strengths for seagrass varied between groups, with local residents giving it a moderately larger outdegree and scientists scoring the indegree twice as high.

Table 4: Category centrality scores by stakeholder group. Centrality is the sum of the indegree and outdegree for each category and is an index of its connectedness to other variables within the map. The categories included below represent the top four categories of each stakeholder group.

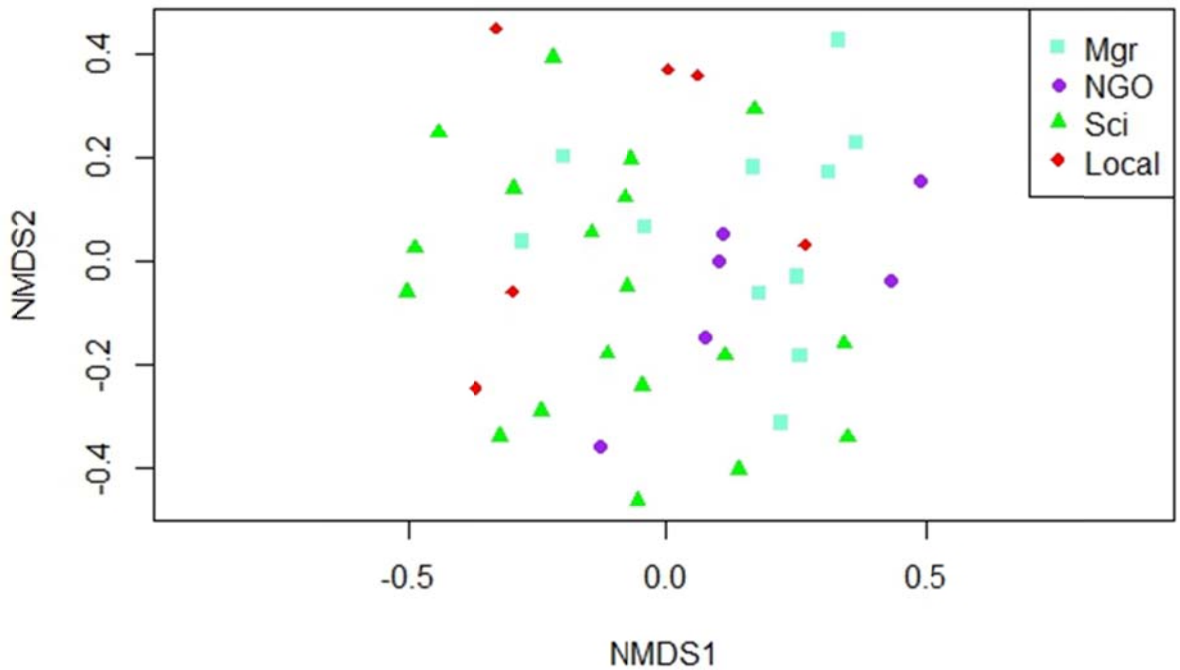
	Scientists	Managers	Environmental NGOs	Local people	Community
Development	1.91	3.93	3.50	3.0	2.75
Human population		3.15		2.48	
Bay ecological condition				2.25	
Seagrass	1.68			1.92	

Bay water quality		3.27	2.75		1.96
Nutrients	3.10		4.25		2.48
Pollution		3.03	3.29		2.00
Fish	1.33				

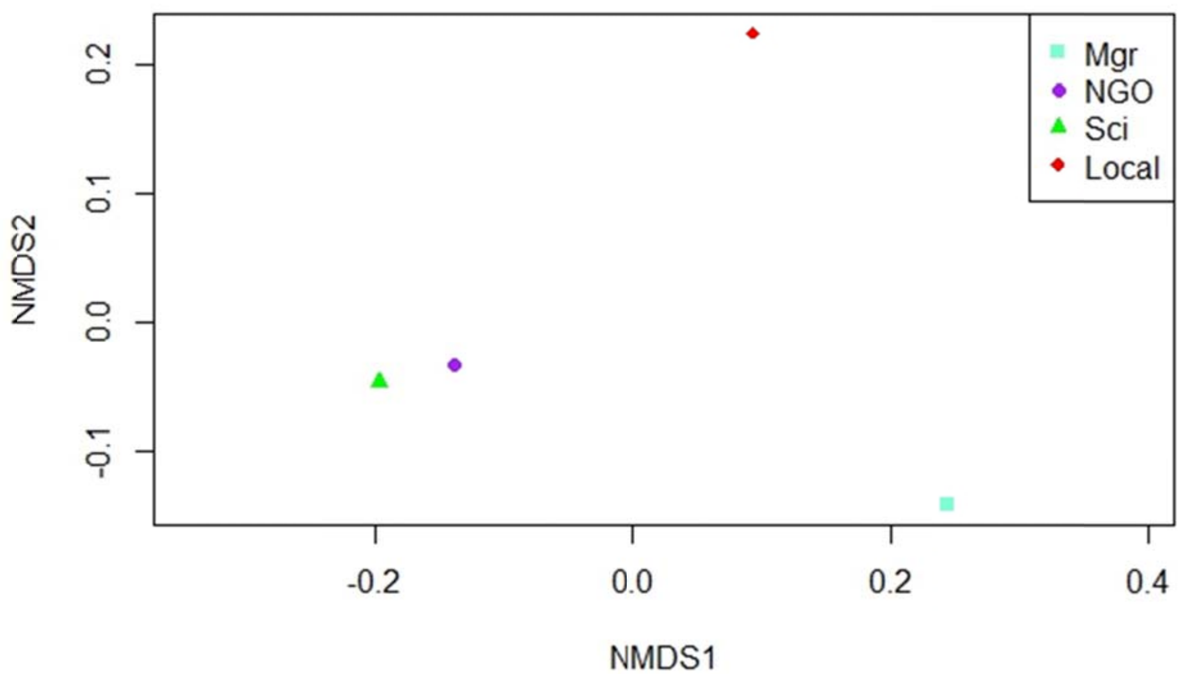
There was substantial overlap in nMDS space between the individual cognitive maps of scientists and all other groups, moderate overlap among managers and NGOs and local residents, and little overlap between NGOs and local residents (Figure 3a). The individuals within each stakeholder group were spread along both nMDS axes, indicating that there is a diversity of conceptual models within each group. When viewed as aggregated stakeholder groups, the Scientist and NGO conceptual models are most similar, while the others are quite dissimilar (Figure 3b).

Figure 3: nMDS plot of the a) individual and b) stakeholder group fuzzy cognitive maps based on centrality scores. Because nMDS is a non-metric procedure, the axes labeled NMDS1 and NMDS2 have no units associated with them. Stress values were 0.279 and 0.169, respectively. Stakeholder groups include Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and Local residents (Local).

a)

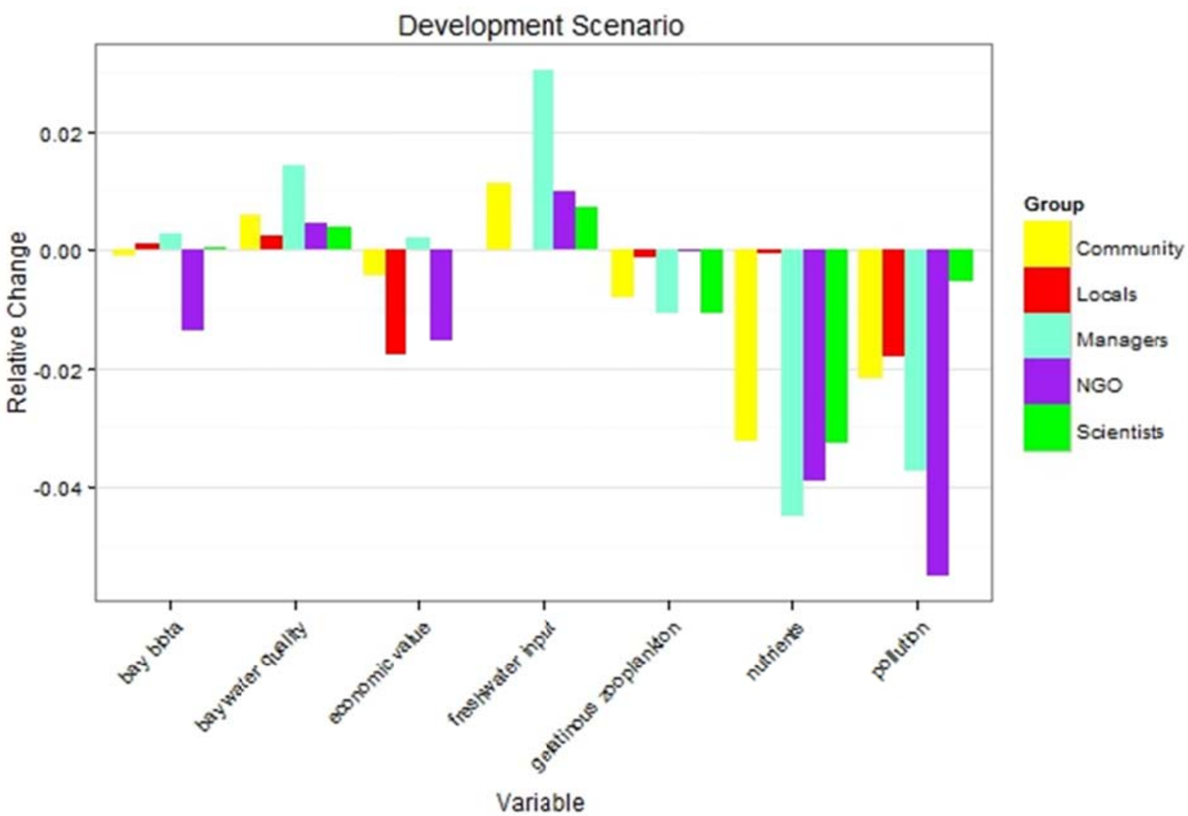


b)



The hypothetical scenario model run further elucidated similarities and differences between the conceptual models of the stakeholder groups (Fig 4). When development was clamped to a low level, nutrients and pollution, two of the more central categories in all groups' models, both decreased compared to the steady state models, though the degree of decline varied among groups. The declines in these two categories were driven primarily by the direct linkages participants made between them and development. The increase in bay water quality and decrease in gelatinous zooplankton (primarily identified by participants as the nuisance jellyfish *Chrysaora quinquecirrha*, or stinging sea nettle) across all groups' models appears to be driven by a number of indirect linkages to development. In the case of bay water quality, one potential pathway identified was a decrease in development leading to a decrease in impervious surfaces, which lead to a decrease in runoff, which improved bay water quality. While the prior examples showed concurrence in the effects of low development across the groups' models, they differed in the outcome of the economic value category; the NGOs' and locals' models predicted a decrease in economic value associated with a decrease in development, while the managers' models predicted an increase in economic value.

Figure 4: Results of the scenario model when development was clamped to a low level. Relative change along the y-axis is the difference between the "low development" scenario compared to the initial steady-state solution for a given category. Stakeholder group models were constructed for Local residents (Local), Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and an aggregate of all cognitive maps (Community).



4.0 Discussion

4.1 The applicability of FCMs in estuarine environments

Fuzzy cognitive maps have been used to model stakeholder perceptions of causal relationships in social-ecological systems in a variety of settings (Özesmi and Özesmi 2003, Meliadou *et al.* 2012, Gray *et al.* 2012, Kontogianni *et al.* 2012, Vanwindekens *et al.* 2013, Zhang *et al.* 2013). This study is the first to apply the methodology to an estuarine ecosystem. Estuaries are both an ecosystem in their own right as well as an ecotone between terrestrial and aquatic and between freshwater and the ocean. Thus, we might expect that people's perceptions of estuaries could be more heterogeneous than FCMs of other systems. The complexity of estuaries is reflected in the large number of unique variables mentioned by the stakeholders during the creation of their FCMs. While caution should be used when comparing FCM indices between studies due to potential differences in methodology (Eden *et al.* 1992), the number of variables recorded in this study exceeds those compiled using similar methods for a large lacustrine system (Özesmi and Özesmi 2003) and a nearshore coastal region (Meliadou *et al.* 2012). This level of detail was not driven by a small number of stakeholders in any particular group; the mean number of categories per map, complexity, and density were all similar across groups, suggesting that all of the stakeholders recognize the complexity and multidimensionality of estuaries.

A potential downside to this is the resulting intricacy of the overall community model, which still includes 84 categories after aggregation. Jørgensen (1994) theorized that quantitative ecological models have a bell-shaped curve in regard to performance verses complexity, and others have suggested that cognitive maps are most easily interpreted when the number of variables ranges from the low teens (Buede and Ferrell 1993) to 30 (Özesmi and Özesmi 2004). Due to its semi-quantitative nature it is difficult to determine how close a FCM approximates the realities of the social–ecological system. However, the models developed here reach a stable state during the scenario analysis in less than 10 iterations and generally follow well established ecological theory, providing additional support for the validity of the findings.

While fuzzy cognitive mapping is robust enough to handle the large number of variables associated with a complex ecosystem, the applicability of this technique is constrained by how well (or poorly) it handles non-monotonic responses (Carvalho 2013). This is particularly true for temperate estuaries, where long gradients in environmental factors like temperature and salinity can lead to dome-shaped response curves. Many of the interviewees attempted to side-step this issue by framing the response in terms of what they anticipated the departure from the current range of the condition would be. For example, interviewees said that increased temperature would lead to an increase in the abundance of a given biota (through some physiological or habitat mediated mechanism) up to some degree, after which increasing temperatures would lead to decreases in abundance. They then posited that it would be unlikely that temperatures in the estuary would ever exceed the inflection point, and thus the overall response is positive. This solution is similar to that previously identified by Hobbs *et al.* (2002) in their construction of an FCM for Lake Erie. Differences in an individual's interpretation on how best to address non-monotonic responses likely led to conflicting causal relationships when aggregating FCMs for the community map. Thus the response of some categories to changes in the scenario model is dampened, though based on notes taken during the interview process it would be limited to a few biotic components and the strength of the interactions tended to be low.

4.2 Differences in stakeholder cognitive models

To develop a comprehensive management plan for complex systems a shared understanding of the components among the stakeholders is a prerequisite (Ogden *et al.* 2005). The findings of this study suggest that while all of the stakeholders interviewed perceive the Barnegat Bay ecosystem as a complex series of social and ecological interconnections and shared common structural elements, there are differences in the components and linkages of their aggregated conceptual models which influence the final state of the system. There is a core set of components that are present in most of the stakeholder groups' FCMs and have high centrality scores; the stakeholder groups all agree that these components are crucial in managing the system towards some desired outcome. However, the number and strength of linkages between these key components and the rest of the social-ecological system varies, such that the FCMs of two stakeholder groups can have opposite outcomes. This was seen in the scenario modeling, where low levels of development through time led to an increase in the economic value of the bay in the Manager's FCM and a decrease in economic value in the NGO and Local models.

One potential reason for the opposing results in the group models may be the primary focus of the groups themselves, including their conception of the relevant "social" dimensions of the system. The individuals comprising the Manager group are tasked with regulating the use of the biological resources of the estuary (fish, crabs, clams, birds), and in their maps a decrease in development yields an increase in biomass and a concomitant increase in economic value through commercial harvest or other recreational opportunities. In contrast, the environmental NGOs often take a broadly anthropocentric view of the social-ecological interactions of the estuary, and their maps contained social and political actors that were not mentioned by others. These social concepts (taxes, land price) often had strongly positive relationships between development and economic value.

While the aggregated community map incorporates multiple perspectives, and thus should be a more complete representation of the system (Gray *et al.* 2012), being able to articulate where, and why, stakeholder groups may have similar or diverging views on important causal relationships will be critical to developing the consensus approach needed to plan appropriate management actions for protection and restoration. A starting point for understanding the convergences or divergences is seen in the arrangement of the group maps in the nMDS, which suggests that the scientists and NGOs place similar importance on a broad variety of categories. This stands in contrast with the managers and local residents, who do not share similar centrality scores among categories. Thus one would expect, and should plan for, the additional effort that will be required to bring these two groups to consensus.

4.3 Further FCM benefits

Opposite interactions (positive versus negative) between two components shared across groups' conceptual models may reflect differences of opinion or perspective but also may point to areas where the understanding of the relationships between concepts is incomplete, such as the effects of climate change on biodiversity and species invasions, and changes to the bay's water quality associated with changes in freshwater input. The identification of these knowledge gaps through FMCs combined with the management objectives developed during the initial stages of the integrated ecosystem assessment will allow for a prioritization of future research and funding needs. These divergences may also indicate subjects where more recent scientific findings have not yet been widely incorporated by those outside specific fields of study (*i.e.* saltmarsh – nutrient interactions, biochemical and physical induced changes in nutrient loads, the pathway and flow of nutrients around the bay) and therefore where additional education/outreach may be warranted. Additionally, the community map can assist in the selection of variables for monitoring once a course of actions has been agreed upon. Given a modeled scenario, or suite of scenarios, the components along the causal chain can be identified, eliminating potential indicators that are not responsive to the management efforts proposed, or do not meet the criteria for informative indicators (Rice and Rochet, 2005). This is particularly important in an age of shrinking research budgets and results-focused management at resource agencies.

5.0 Conclusion

We have shown that Fuzzy Cognitive Mapping can be a useful tool for organizing the intricate connections between social and ecological concepts within a highly complex ecosystem, and when applied across stakeholder groups can elucidate not only those mechanisms for which there is a shared understanding, but also highlight where additional resources should be focused to gain the greatest insights into system operation. While subject to limitations associated with representing non-monotonic response variables, they can nevertheless serve as a basis from which the initial steps of an Integrated Ecosystem Assessment can proceed. In particular, the individual interview procedure utilized herein avoids some of the pitfalls associated with group participation in the scoping process and provides a clear scaffolding upon which potential management and policy scenarios can be evaluated.

6.0 Acknowledgements

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7.0 References

An, L. and Lopez-Carr, D. 2012. Understanding human decisions in coupled natural and human systems. Ecological Modelling 229:1-4.

Axelrod, R., 1976. Structure of Decision: The Cognitive Maps of Political Elites. Princeton University Press, Princeton, NJ, USA.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. Effects of Nutrient Enrichment In the Nation's Estuaries: A Decade of Change. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD. 328 pp.

Bougon, M., Weick, K., Binkhorst, D., 1977. Cognition in organizations: an analysis of the Utrecht Jazz Orchestra. Administrative Science Quarterly. 22, 606–639.

Buede, D.M., Ferrell, D.O., 1993. Convergence in problem solving: a prelude to quantitative analysis. IEEE Transactions On Systems Man and Cybernetics 23, 746–765.

Carley, K., Palmquist, M., 1992. Extracting, representing, and analyzing mental models. Social Forces 70, 601–636.

Carvalho, J.P. 2013. On the semantics and the use of fuzzy cognitive maps and dynamic cognitive maps in social sciences. Fuzzy Sets and Systems. 214:6-19.

Churchman, C.W. 1967. Wicked Problems. Management Science. 14(4) B141-142.

Clarke, K.R., and R.M. Warwick. 2001. Change in marine communities: An approach to statistical analysis and interpretation, 2nd ed. Plymouth: PRIMER-E.
Dickerson, J.A. and Kosko, B. 1994 Virtual Worlds as Fuzzy Cognitive Maps. Presence. 3(2): 173-189.

Eden, C., Ackerman, F., Cropper, S., 1992. The analysis of cause maps. Journal of Management Studies 29, 309–323.

Gray, S., Chan, A., Clark, D., Jordan, R. 2012. Modeling the integration of stakeholder knowledge in social–ecological decision-making: Benefits and limitations to knowledge diversity. Ecological Modeling. 229:88-96

Hage, P., Harary, F., 1983. Structural Models in Anthropology. Oxford University Press, New York.

Harary, F., Norman, R.Z., Cartwright, D., 1965. Structural Models: An Introduction to the Theory of Directed Graphs. John Wiley & Sons, New York.

Hobbs, B.F., Ludsin, S.A., Knight, R.L., Ryan, P.A., Biberhofer, J., Ciborowski, J.J.H., 2002. Fuzzy Cognitive Mapping as a Tool to Define Management Objectives for Complex Ecosystems. Ecological Applications 12, 1548-1565.

Jentoft, S. and Chuenpagdee, R. 2009. Fisheries and coastal governance as a wicked problem. Marine Policy 33:553–560

Jørgensen, S.E., 1994. Fundamentals of Ecological Modelling. Elsevier, New York, 628 pp.

Kennish, M.J. 2001. Physical description of the Barnegat Bay – Little Egg Harbor estuarine system. Journal of Coastal Research Special Issue 32: 13-27

Kontogianni, A., Papageorgiou, E., Salomatina, L., Skourtos, M., and Zanou, B. 2012. Risks for the Black Sea marine environment as perceived by Ukrainian stakeholders: A fuzzy cognitive mapping application. Ocean & Coastal Management. 62:34-42.

Kontogianni, A., Papageorgiou, E., and Tourkolias, C. 2012b. How do you perceive environmental change? Fuzzy Cognitive Mapping informing stakeholder analysis for environmental policy making and non-market valuation. Applied Soft Computing, 12:3725-3735.

Kosko, B., 1986. Fuzzy cognitive maps. International Journal of Man–Machine Studies. 1, 65–75.

Kosko, B., 1987. Adaptive inference in fuzzy knowledge networks. In: Proceedings of the First IEEE International Conference on Neural Networks (ICNN-86), San Diego, CA, pp. 261–268.

Kosko, B., 1991. Neural Networks and Fuzzy Systems. Prentice-Hall, Englewood Cliffs, NJ, USA.

Levin, P.S., M.J. Fogarty, G.C. Matlock, and M. Ernst. 2008. Integrated ecosystem assessments. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-NWFSC-92, 20 p.

Levin, P.S., Fogarty, M.J., Murawski, S.A., Fluharty, D., 2009. Integrated Ecosystem Assessments: Developing the Scientific Basis for Ecosystem-Based Management of the Ocean. PLoS Biology 7, 0023-0028.

Liu, J., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C.L., Schneider, S.H., Taylor, W.W., 2007. Complexity of coupled human and natural systems. Science 317, 1513–1516.

Ludwig, D. 2001. The Era of Management Is Over. Ecosystems 4: 758–764.

McClure M, Ruckelshaus M. 2007. Collaborative science: Moving ecosystem-based management forward in Puget Sound. Fisheries 32: 458.

Meliadou, A., Santoro, F., Nader, M.R., Dagher, M.A., Indary, S.A., Salloum, B.A. 2012. Prioritising coastal zone management issues through fuzzy cognitive mapping approach. Journal of Environmental Management. 97:56-68.

National Oceanic and Atmospheric Administration. 2006. Evolving an ecosystem approach to science and management through NOAA and its partners. Available: http://www.sab.noaa.gov/Reports/eETT_Final_1006.pdf. Accessed 6 January 2014.

National Research Council. 2008. *Public Participation in Environmental Assessment and Decision Making*. Panel on Public Participation in Environmental Assessment and Decision Making, Thomas Dietz and Paul C. Stern, eds. Committee on the Human Dimensions of Global Change. Division of Behavioral and Social Sciences and Education. Washington, DC: The National Academies Press.

Ogden, J.C., Davis, S.M., Jacobs, K.J., Barnes, T., Fling, H.E., 2005. The Use of Conceptual Ecological Models to Guide Ecosystem Restoration in South Florida. Wetlands 25, 795-809.

Özesmi, U., Özesmi, S.L., 2003. A Participatory Approach to Ecosystem Conservation: Fuzzy Cognitive Maps and Stakeholder Group Analysis in Uluabat Lake, Turkey. Environmental Management 31, 518-531.

Özesmi, U., Özesmi, S.L., 2004. Ecological models based on people's knowledge: a multi-step fuzzy cognitive mapping approach. Ecological Modelling 176, 43-64.

Raymond, C.M., Fazey, J., Reed, M.S., Stringer, L.C., Robinson, G.M., Evely, A.C., 2010. Integrating local and scientific knowledge for environmental management. Journal of Environmental Management 91, 1766-1777.

Rice, J.C., Rochet, M.-J., 2005. A framework for selecting a suite of indicators for fisheries management. ICES Journal of Marine Science, 62, 516-527.

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, JL., Sheilb,D., Meijaard, E., Venter, M., Boedhihartono, A.K., Day, M., Garcia, C., van Oosten C., and Buck, L.E. 2013. Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. Proceedings of the National Academy of Science 110(21): 8349–8356.

United States Census Bureau. 2012. State and County QuickFacts. http://quickfacts.census.gov/qfd/states/34/34029.html. Accessed 09, January 2014.

Vanwindekens, F.M., Stilmant, D., Baret, P.V. 2013. Development of a broadened cognitive mapping approach for analysing systems of practices in social–ecological systems. Ecological Modelling. 250:352– 362

Xiang, W. 2013. Working with wicked problems in socio-ecological systems: Awareness, acceptance, and adaptation. Landscape and Urban Planning. 110(1-4).

Zhang, H., Song, J., Su, C., He, M. 2013. Human attitudes in environmental management: Fuzzy Cognitive Maps and policy option simulations analysis for a coal-mine ecosystem in China. Journal of Environmental Management. 115:227-234.

8.0 Supplemental Information

Table S1: Centrality scores by stakeholder group cognitive models. A blank value indicates a category not included in that particular group's model. The Community model is the aggregate of all individual models.

Category	Scientist	Manager	NGO	Local residents	Community
agriculture	0.34	0.20	0.08		0.22
algal blooms	0.25	0.18	0.54	0.25	0.27
atmospheric deposition	0.43	0.64	1.12		0.43
bay biota	0.61	1.32	2.35	0.71	1.04
bay ecological condition	0.30	1.02	0.50	2.25	0.71
bay salinity	0.99	0.57	1.48	0.38	0.82
bay water quality	1.04	3.27	2.75	1.88	1.96
bay water temperature	0.78	0.80	1.92	0.42	0.71
benthic biota	0.96			0.25	0.47
benthic infauna	0.41				0.19
biochemical/physical processes	0.86		0.17	0.13	0.41
biodiversity	0.12	0.20	0.25		0.11
birds	0.20	0.09	0.54	0.79	0.30
blue crabs	0.33	0.34	0.50	0.54	0.39
boating	0.91	0.70	1.04	1.27	0.88
bulkheading/docks	0.57	0.86	0.71	0.71	0.61
climate change	0.59	1.07	1.37		0.71
commercial fishing	0.28	1.10	0.13	0.13	0.44
conservation	0.03	0.77	0.13	0.88	0.29
depth	0.24	0.07	0.50	0.25	0.16
development	1.91	3.93	3.50	3.00	2.75
dissolved oxygen	0.80	0.33	0.79	0.75	0.63
dredging	0.20		0.25	0.25	0.16
economic value	0.37	1.49	0.88	0.50	0.75
ecosystem services		0.68	0.21		0.21
effective management	0.24	0.78	2.16		0.62
elected officials			1.24	0.50	0.25
erosion	0.28	0.18	0.54	0.25	0.27
fish	1.33	1.39	1.54	1.50	1.33
fishing	0.58	1.02	1.75	0.38	0.81
freshwater input	1.13	2.44	2.15	0.13	1.34
freshwater quality	0.33	0.72	1.33	0.75	0.61
freshwater use	0.50	1.07	1.42	0.38	0.71
gelatinous zooplankton	1.05	0.39	1.33	0.63	0.86

Table S1: Centrality scores by stakeholder group cognitive models. A blank value indicates a category not included in that particular group's model. The Community model is the aggregate of all individual models.					
Category	Scientist	Manager	NGO	Local residents	Community
geomorphological processes	0.29	0.47	0.17		0.27
government	0.04	0.60	1.46	0.38	0.34
hard clams	0.38	0.66	0.38	0.50	0.47
harmful algal blooms	0.45	0.32	0.25		0.31
household inputs	0.30	0.39	0.25	1.00	0.42
human population	0.88	3.15	1.50	2.48	1.74
impervious surfaces	0.22	1.09	1.96		0.67
intangible values	0.17	0.86	0.42	0.38	0.38
invasive species	0.18	0.51		0.29	0.25
larval supply	0.50	0.32	0.17		0.33
macroalgae	0.18	0.11	0.46	0.88	0.30
microbial loop	0.41		0.33		0.23
natural habitat	0.99	1.64	1.27	0.38	1.08
NGOs			1.19	0.54	0.25
nutrients	3.10	2.10	4.25	0.63	2.48
ocean exchange	1.31	1.18	1.63	0.25	1.00
OCNGS	0.49	0.66	1.83	0.08	0.60
other crustaceans		0.18		1.13	0.21
other groups		0.36	0.34		0.12
other land use	0.58	0.84	1.33	0.38	0.68
other plankton	0.22		0.54	0.25	0.21
other recreational use	1.25	1.62	1.00	1.88	1.32
oysters	0.16		0.29	0.38	0.17
phytoplankton	1.27	0.40			0.64
policy decisions	0.13	1.50	0.46	0.13	0.47
pollution	1.32	3.03	3.29	1.63	2.00
precipitation	0.16	0.12	0.46		0.17
preserved open space	0.33	1.30	1.04	0.50	0.71
public	0.17	0.41	1.04		0.33
public awareness	0.20	0.91	1.08	1.58	0.68
recreational fishing	0.28	0.68		0.13	0.27
regulations	0.30	0.30	0.63	0.25	0.32
residence time	0.59	0.98	0.58		0.58
resource users	0.04		1.92		0.29
runoff	0.53	0.39	1.17	0.63	0.60
salt marshes	0.59	0.59	0.17	0.38	0.48
scientists			1.33		0.19
seagrass	1.68	1.00	1.17	1.92	1.46
sediment	0.73	0.34			0.42

Table S1: Centrality scores by stakeholder group cognitive models. A blank value indicates a category not included in that particular group's model. The Community model is the aggregate of all individual models.

Category	Scientist	Manager	NGO	Local residents	Community
sewer systems	0.07	0.08	1.08	0.63	0.26
shellfish	0.70	0.66	0.88	0.71	0.67
stormwater	0.12	0.57	0.13	0.13	0.24
tides	0.33	0.32	0.54		0.27
tourism	0.09	1.23	1.88	0.25	0.67
turbidity	0.93	0.18	0.54	0.50	0.62
vehicles	0.07	0.50	0.67	0.42	0.32
water circulation	0.74	0.30	0.25	0.25	0.41
wetlands	0.04	1.03	0.21		0.32
wind	0.13		0.29	0.13	0.12
zooplankton	0.64	0.16	0.38	0.25	0.42

Figure S1. Conceptual schematic of the FCM combination process and steady state calculation.

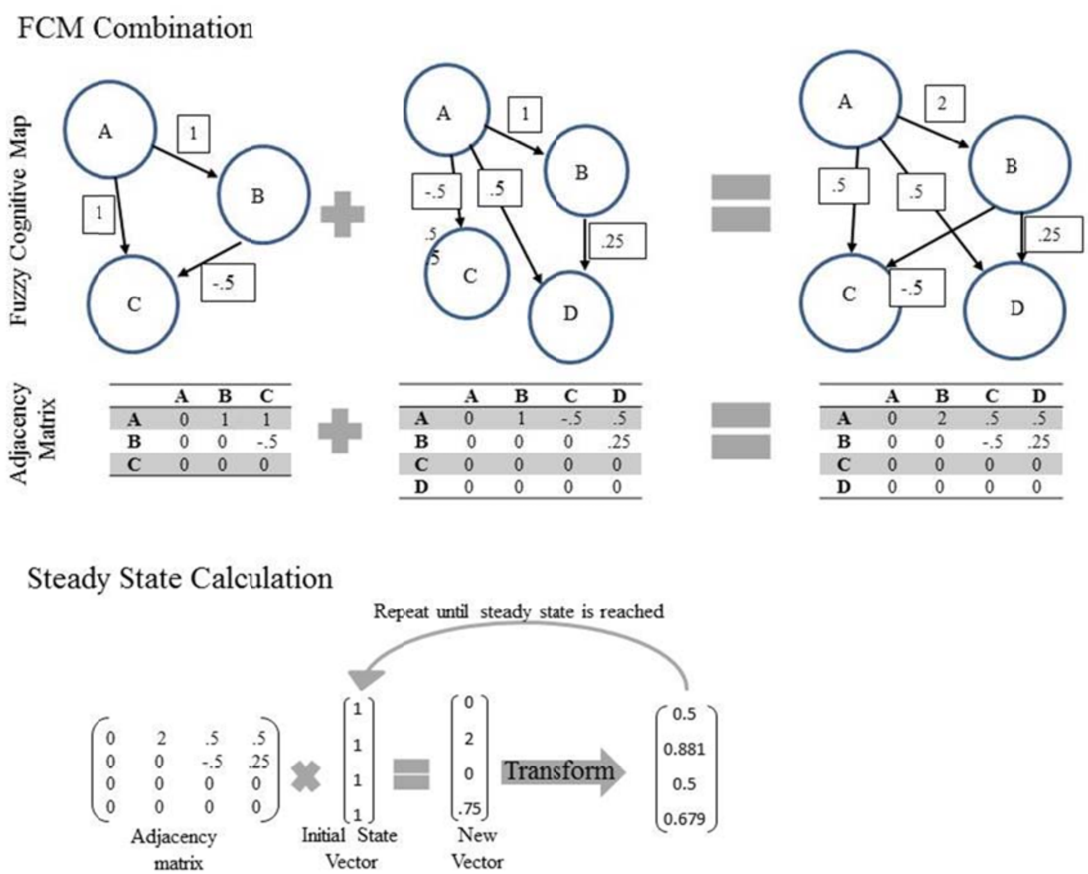

Figure S2. A flow diagram of the data analysis steps.

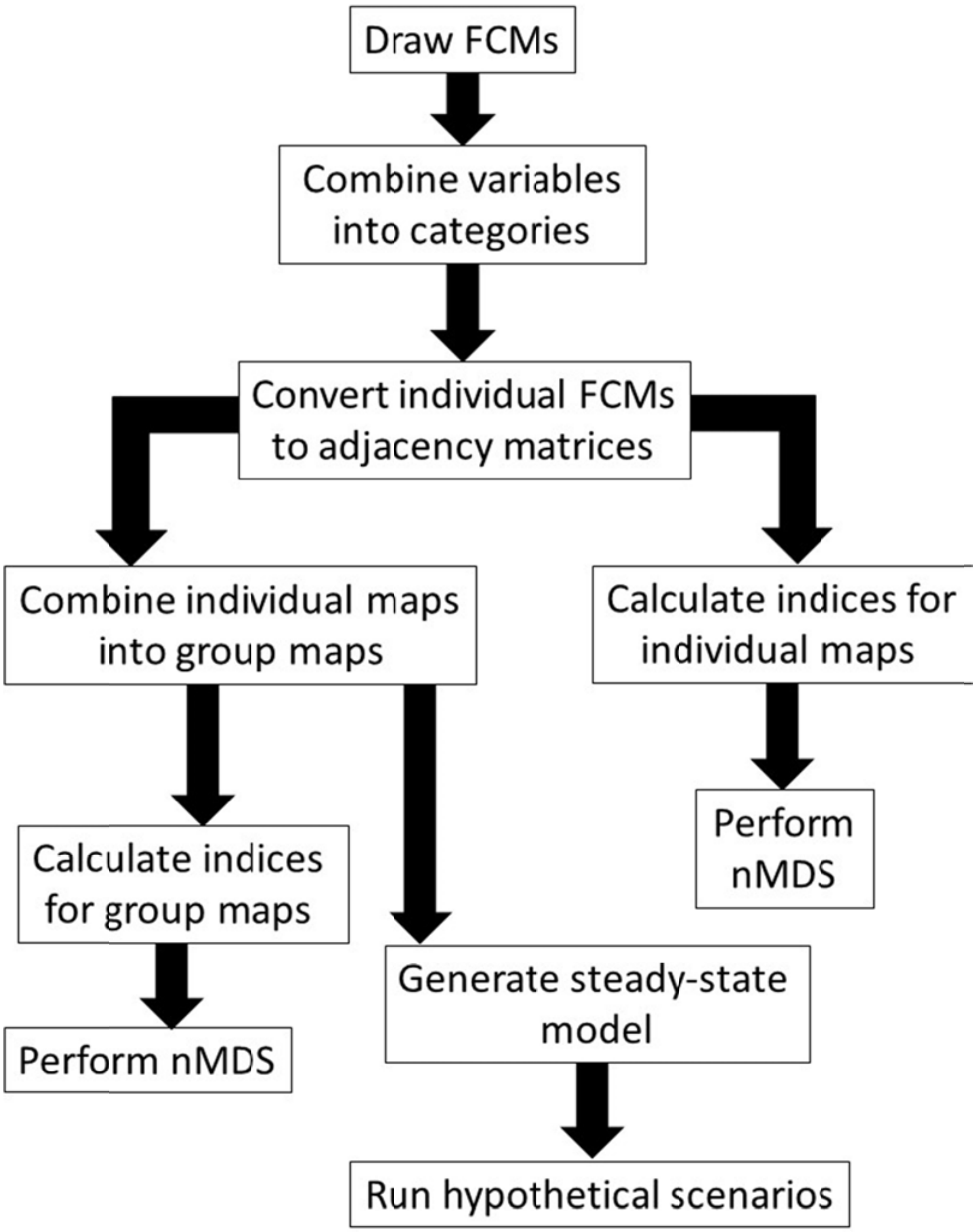

Figure S3. Accumulation curves for the total number of categories versus the number of interviews. The black line is scientists, red is managers, blue is local people, and green in environmental NGOs.

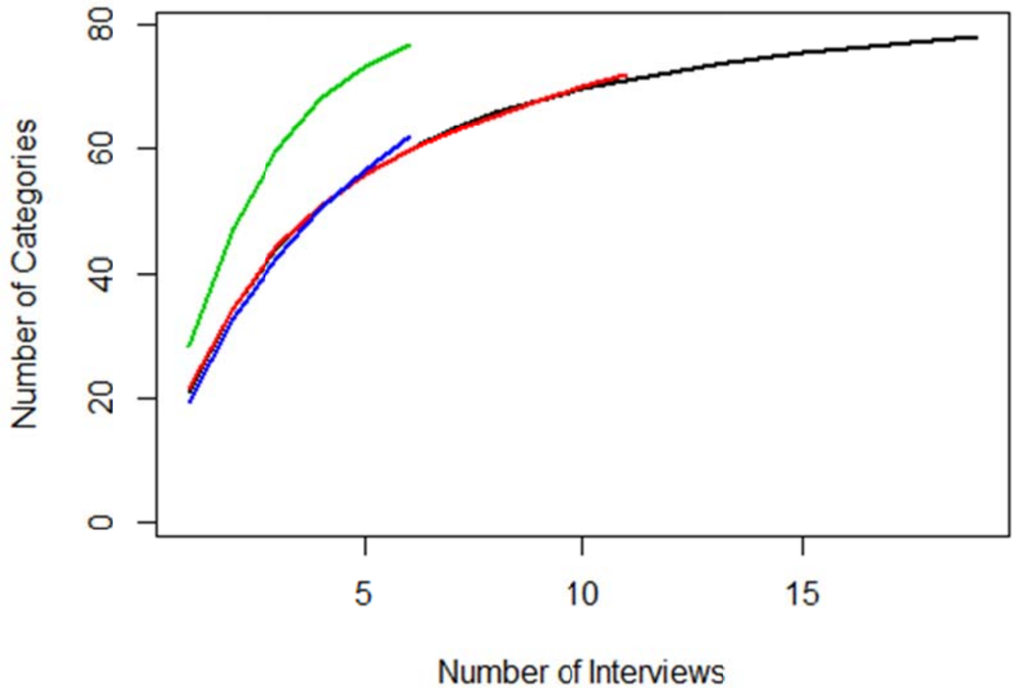

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Figure 1 – Map of Barnegat Bay watershed with New Jersey inset.

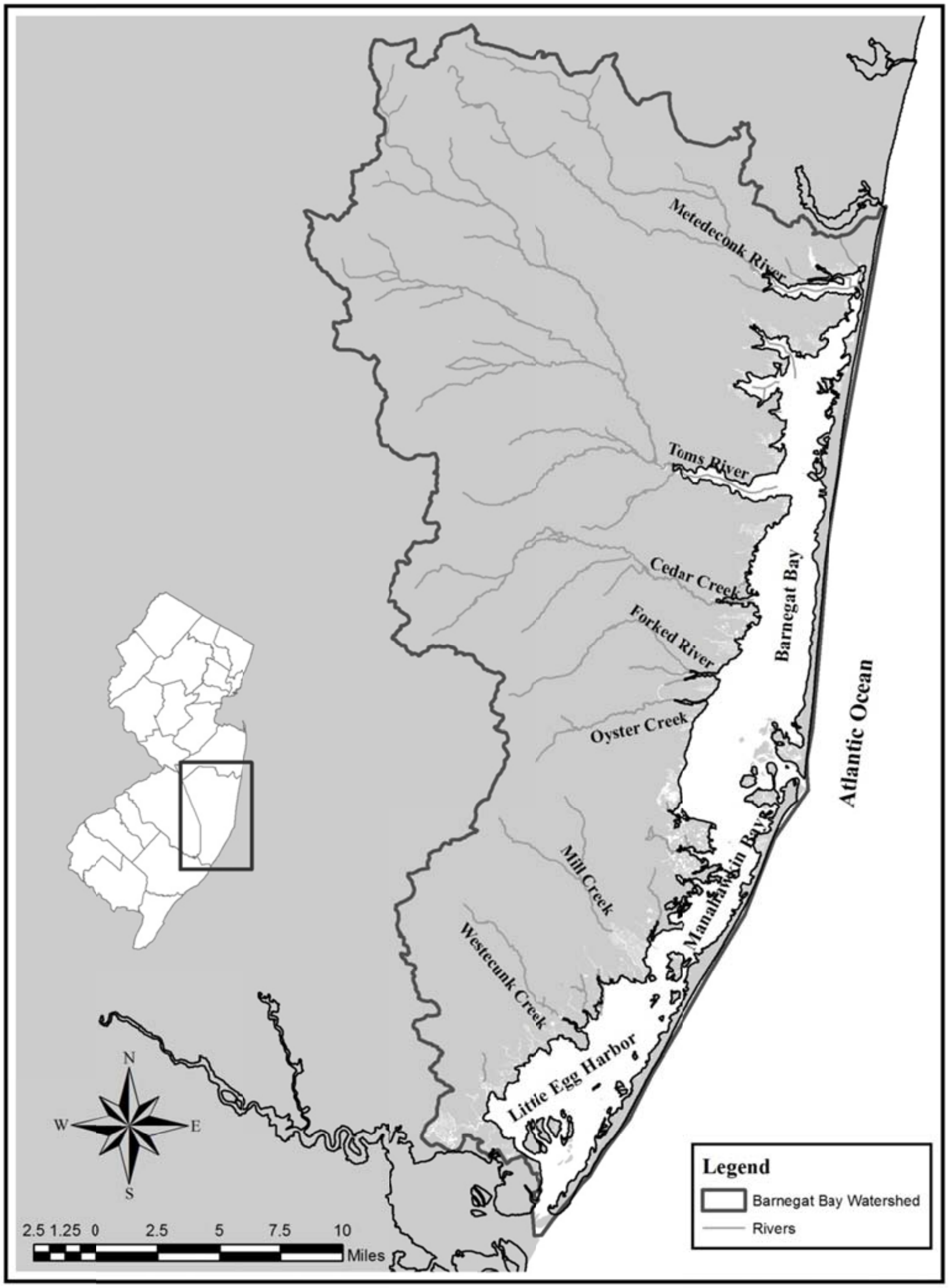


Figure 2: Subset of the community conceptual model. The twelve nodes with centrality scores greater than 1.0 are shown. Node size is related to centrality score, solid lines are positive interaction strengths, dotted lines are negative interactions strengths.

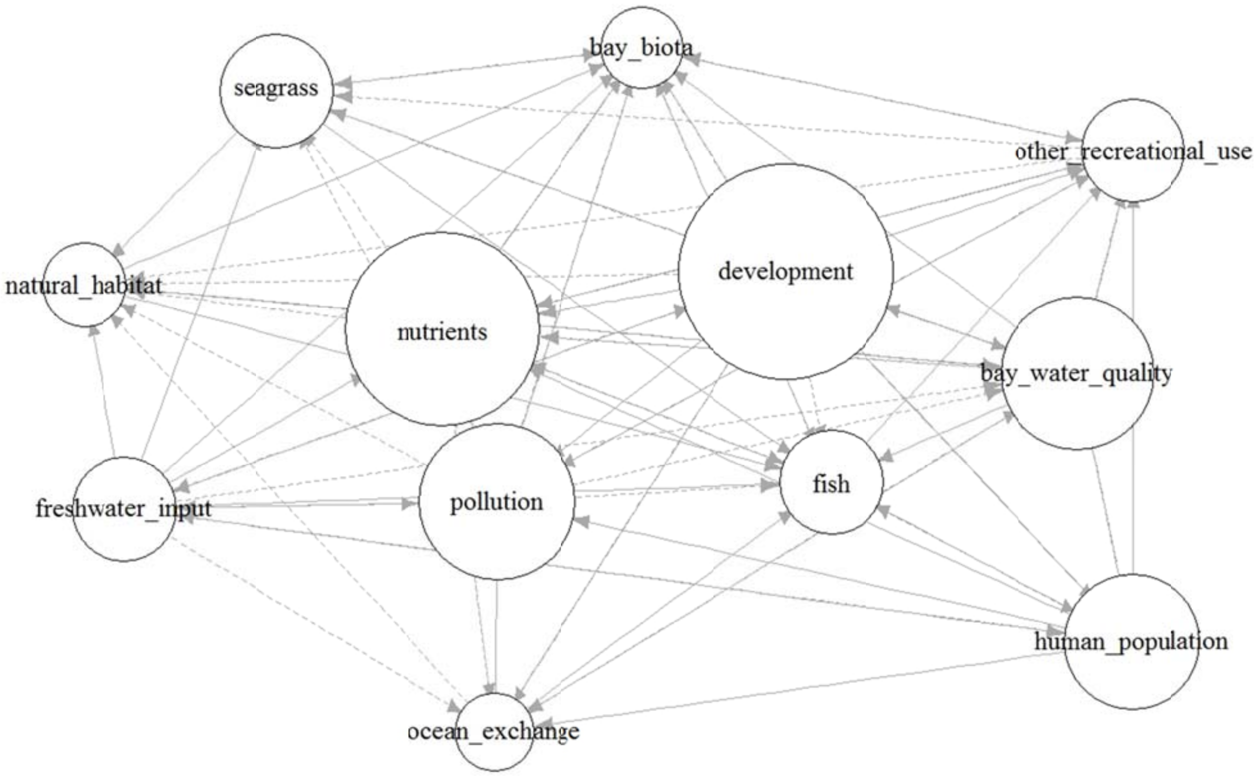
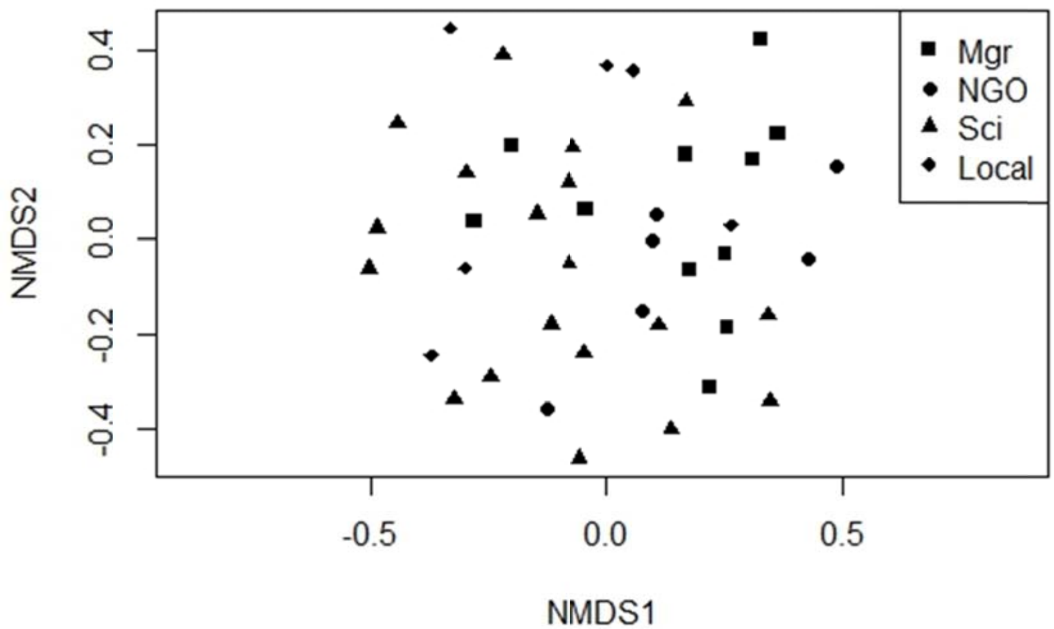


Figure 3: nMDS plot of the a) individual and b) stakeholder group fuzzy cognitive maps based on centrality scores. Stress values were 0.279 and 0.169, respectively. Stakeholder groups include Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and Local residents (Local).

a)



b)

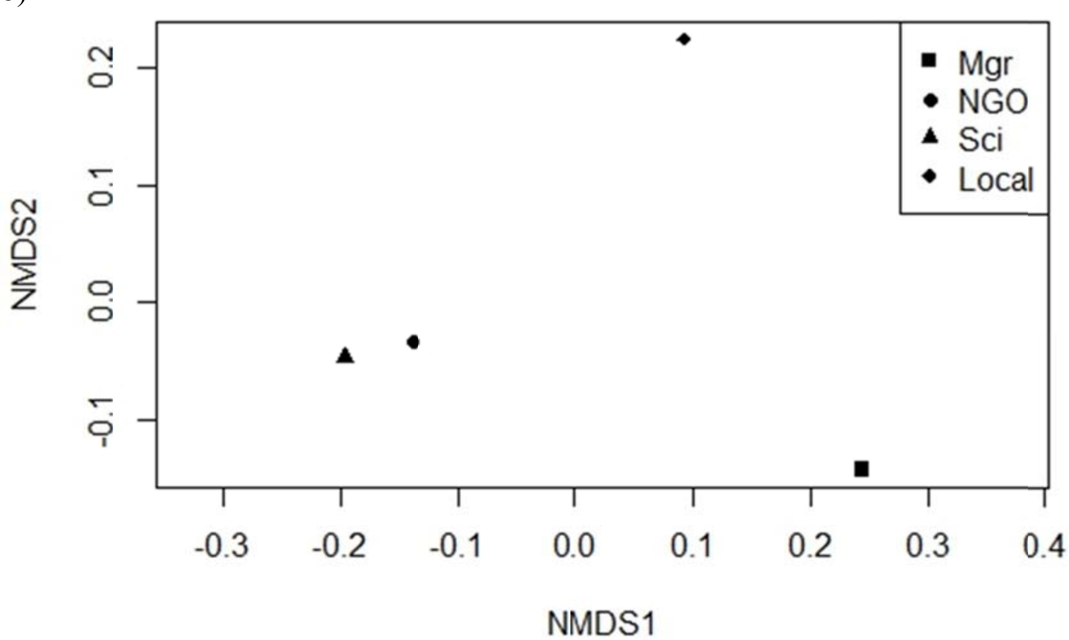


Figure 4: Results of the scenario model when development was clamped to a low level. Relative change along the y-axis is the difference between the "low development" scenario compared to the initial steady-state solution for a given category. Stakeholder group models were constructed for Local residents (Local), Managers (Mgr), Environmental non-governmental organizations (NGO), Scientists (Sci), and an aggregate of all cognitive maps (Community).